

REMEDIAL INVESTIGATION REPORT
SILVER MOUNTAIN MINE
OKANOGAN COUNTY, WASHINGTON

January 19, 1990

U.S. Environmental Protection Agency
Region 10
Seattle, Washington

1574

AR 3.7 0001

RI TABLE OF CONTENTS
PAGE 1

REMEDIAL INVESTIGATION TABLE OF CONTENTS

EXECUTIVE SUMMARY

LIST OF FIGURES

LIST OF TABLES

ABBREVIATIONS OF CHEMICAL AND QUANTITATIVE TERMS

CHAPTER 1. INTRODUCTION PAGE

1.1 PURPOSE AND SCOPE OF REPORT.....	1
1.2 SITE BACKGROUND.....	2
1.2.1 SITE DESCRIPTION.....	3
1.2.2 SITE HISTORY AND LAND USE.....	5
1.2.3 REGULATORY HISTORY.....	7
1.2.4 PREVIOUS INVESTIGATIONS.....	12
1.3 REPORT ORGANIZATION.....	13
REFERENCES.....	14

CHAPTER 2. STUDY AREA INVESTIGATION

2.1 SURFACE FEATURES.....	1
2.2 GEOLOGICAL INVESTIGATION.....	1
2.3 ANALYTICAL PARAMETERS.....	4
2.4 LEACH HEAP AND MINE DUMP INVESTIGATION.....	10
2.4.1 LEACH HEAP AND MINE DUMP SAMPLING PROCEDURES...	10
2.4.2 LEACH HEAP AND MINE DUMP PHYSICAL PARAMETERS...	14
2.5 SOILS INVESTIGATION.....	15
2.5.1 CYANIDE AND METALS ANALYSIS.....	15
2.5.2 ORGANICS ANALYSIS.....	16
2.6 METEOROLOGICAL INVESTIGATION.....	18
2.7 SURFACE WATER INVESTIGATION.....	19
2.8 GROUND WATER INVESTIGATION.....	21
2.8.1 OFFSITE WELLS.....	21
2.8.2 ONSITE MONITORING WELLS.....	23
2.8.2.1 MONITORING WELL INSTALLATION.....	23
2.8.2.2 SLUG TEST.....	24
2.9 WATER SAMPLING PROCEDURES.....	24
2.9.1 STATIC WATER LEVELS.....	24
2.9.2 TEMPERATURE.....	29
2.9.3 EH, PH, ELECTRICAL CONDUCTIVITY.....	29
2.9.4 SAMPLE COLLECTION AND PRESERVATION.....	29
2.10 DEMOGRAPHIC INVESTIGATION.....	31
2.11 ECOLOGICAL INVESTIGATION.....	31
REFERENCES.....	31

CHAPTER 3. PHYSICAL CHARACTERISTICS OF THE STUDY AREA

RI TABLE OF CONTENTS

PAGE 2

3.1	TOPOGRAPHY.....	1
3.2	CLIMATE.....	1
3.3	GEOLOGY.....	2
3.3.1	REGIONAL GEOLOGY.....	2
3.3.2	LOCAL GEOLOGY.....	2
3.4	SOILS AND LEACH HEAP MATERIALS.....	3
3.5	HYDROLOGY AND HYDROGEOLOGY.....	8
3.5.1	SURFACE WATER HYDROLOGY.....	8
3.5.2	SITE AQUIFER.....	9
3.5.2.1	AQUIFER THICKNESS.....	9
3.5.2.2	STATIC WATER LEVELS.....	11
3.5.2.3	HYDRAULIC CONDUCTIVITY AND FLOW VELOCITY.....	11
3.5.2.4	FLOW DIRECTION.....	14
3.5.2.5	GROUND WATER BUDGET.....	14
3.5.2.6	GROUND WATER TEMPERATURE.....	15
3.5.3	FIELD RESULTS.....	21
3.5.4	SHALLOW AQUIFER SUMMARY.....	23
3.6	ECOLOGY.....	23
	REFERENCES.....	24

CHAPTER 4. NATURE AND EXTENT OF CONTAMINATION

4.1	LEACH HEAP AND MINE DUMP.....	1
4.1.1	LOCATION OF SAMPLE SITES.....	1
4.1.2	ANALYTICAL RESULTS.....	1
4.2	EXTENT OF SOIL CONTAMINATION.....	8
4.2.1	LOCATION OF SAMPLE SITES.....	8
4.2.2	SOILS ANALYTICAL RESULTS.....	8
4.2.2.1	INORGANIC RESULTS.....	8
4.2.2.2	ORGANIC RESULTS.....	9
4.3	EXTENT OF WATER CONTAMINATION.....	13
4.3.1	LOCATION OF SAMPLE SITES.....	13
4.3.2	ANALYTICAL RESULTS.....	13
4.3.2.1	ONSITE GROUND WATER.....	14
4.3.2.2	ONSITE SURFACE WATER.....	15
4.3.2.3	OFFSITE GROUND AND SURFACE WATER.....	15
4.4	NATURE OF CONTAMINATION.....	26
4.4.1	CYANIDE.....	26
4.4.1.1	CYANIDE IN ROCK AND SOIL.....	26
4.4.1.2	CYANIDE IN WATER.....	28
4.4.2	ARSENIC.....	30
4.4.2.1	ARSENIC IN ROCK AND SOIL.....	30
4.4.2.2	ARSENIC IN WATER.....	31
4.4.3	SOURCES OF CONTAMINANTS IN WATER.....	31
4.5	SUMMARY.....	32
	REFERENCES.....	33

CHAPTER 5. FATE AND TRANSPORT OF CONTAMINANTS

RI TABLE OF CONTENTS
PAGE 3

5.1	CONCEPTUAL MODEL OF HYDROGEOLOGIC SYSTEM.....	1
5.2	WATER TRANSPORT.....	4
5.2.1	INFILTRATION AND UNSATURATED FLOW.....	4
5.2.2	GROUND WATER FLOW.....	6
5.2.3	UNCERTAINTY IN WATER TRANSPORT ESTIMATES.....	6
5.3	CHEMICAL TRANSPORT.....	7
5.3.1	CONTAMINANTS OF CONCERN.....	7
5.3.2	PROBABLE FATE OF CYANIDE.....	8
5.3.2.1	SPECIATION OF CYANIDE.....	9
5.3.2.2	SOLUBILITY LIMITS OF CYANIDE.....	11
5.3.3	PROBABLE FATE OF ARSENIC.....	14
5.3.3.1	SPECIATION OF ARSENIC.....	14
5.3.3.2	SOLUBILITY LIMITS OF ARSENIC UNDER OXIDIZING CONDITIONS.....	15
5.3.4	TRANSPORT OF ARSENIC AND CYANIDE.....	18
5.3.4.1	CYANIDE DEGRADATION.....	18
5.3.4.2	ARSENIC ADSORPTION.....	21
5.3.4.3	PROJECTED GROUND WATER CONCENTRATIONS..	21
5.3.5	UNCERTAINTY IN CONTAMINANT TRANSPORT ESTIMATES.....	22
5.5	SUMMARY.....	23
	REFERENCES.....	25

CHAPTER 6. HUMAN HEALTH RISK ASSESSMENT

6.1	CONTAMINANT IDENTIFICATION.....	1
6.2	TOXICITY ASSESSMENT.....	4
6.2.1	NONCARCINOGENIC EFFECTS.....	5
6.2.2	CARCINOGENIC EFFECTS.....	6
6.2.3	CHEMICALS OF SPECIAL CONCERN.....	8
6.2.3.1	ARSENIC.....	8
6.2.3.2	LEAD.....	8
6.2.3.3	NITRATE/NITRITE.....	10
6.2.3.4	COPPER.....	10
6.2.4	DERMAL CONTACT.....	10
6.2.5	UNCERTAINTY.....	11
6.3	EXPOSURE ASSESSMENT.....	12
6.3.1	POTENTIALLY EXPOSED POPULATIONS.....	12
6.3.2	REASONABLE MAXIMUM EXPOSURE.....	13
6.3.3	EXPOSURE PATHWAYS AND ROUTES.....	14
6.3.4	EXPOSURE POINT CONCENTRATIONS.....	17
6.3.4.1	WATER.....	17
6.3.4.2	SOIL.....	18
6.3.5	CONTAMINANT FATE/TRANSPORT.....	18
6.3.5.1	AIR FATE/TRANSPORT.....	18
6.3.5.2.1	PARTICULATE EMISSIONS.....	18
6.3.5.2.2	VOLATILIZATION OF CYANIDE.....	20
6.3.5.2	WATER FATE/TRANSPORT.....	21
6.3.5.3	SOIL FATE/TRANSPORT.....	21

RI TABLE OF CONTENTS
PAGE 4

6.3.6	ESTIMATE CHEMICAL INTAKE.....	21
6.4	RISK CHARACTERIZATION.....	24
6.4.1	DEFINITION OF RISK ENDPOINTS.....	24
6.4.1.1	CARCINOGENIC EFFECTS.....	24
6.4.1.2	NONCARCINOGENIC EFFECTS.....	24
6.4.2	CARCINOGENIC RISK.....	26
6.4.2.1	RISK BY PATHWAY.....	26
6.4.2.2	RISK BY MEDIUM.....	27
6.4.2.3	COMBINED CARCINOGENIC RISK.....	27
6.4.2.4	STOCK TANK.....	28
6.4.3	NONCARCINOGENIC RISK.....	28
6.4.3.1	RISK BY CHEMICAL.....	28
6.4.3.2	RISK BY MEDIUM.....	29
6.4.3.3	STOCK TANK.....	29
6.4.3.4	LEAD.....	29
6.4.3.5	SUMMARIZING BY CRITICAL EFFECT.....	34
6.5	UNCERTAINTY ASSESSMENT.....	34
6.5.1	TOXICITY.....	38
6.5.2	EXPOSURE.....	38
6.6	SUMMARY.....	39
	REFERENCES.....	40

CHAPTER 7. BASELINE ENVIRONMENTAL RISK ASSESSMENT

7.1	CONTAMINANT IDENTIFICATION.....	1
7.2	TOXICITY ASSESSMENT.....	1
7.2.1	INITIAL SCREENING.....	1
7.2.1.1	EVALUATION OF SURFACE AND GROUND WATER DATA.....	1
7.2.1.2	EVALUATION OF SOIL DATA.....	5
7.2.2	TOXICITY OF SURFACE AND GROUND WATER.....	6
7.2.3	FUTURE TOXICITY OF CYANIDE AND ARSENIC IN GROUND WATER.....	7
7.2.4	TOXICITY OF SOIL.....	7
7.2.5	TOXICITY TESTS AND THEIR QUALITY.....	14
7.2.6	UNCERTAINTY.....	14
7.3	EXPOSURE ASSESSMENT.....	16
7.3.1	POTENTIALLY EXPOSED POPULATIONS.....	16
7.3.1.1	ECOSYSTEM CHARACTERISTICS.....	16
7.3.1.2	SENSITIVE ECOSYSTEMS.....	20
7.3.1.3	ECOLOGICAL ENDPOINTS.....	22
7.3.2	EXPOSURE PATHWAYS.....	22
7.3.3	EXPOSURE POINT CONCENTRATIONS.....	23
7.3.4	SITE MODEL.....	23
7.3.5	EXPOSURE SUMMARY.....	25
7.3.6	UNCERTAINTY.....	25
7.4	RISK CHARACTERIZATION.....	25
7.5	SUMMARY.....	26
	REFERENCES.....	28

APPENDICES

- A. HEAP LEACHING METHODS
- B. PHOTOGRAPHS OF SITE
- C. FIELD NOTES FROM 1988-1989
- D. CHEMICAL ANALYSES AND DATA QUALITY VALIDATION
- E. PETROGRAPHIC ANALYSIS
- F. LIST OF UNITS USED IN REPORT
- G. REGION 10 EXPOSURE PARAMETERS AND JUSTIFICATION
- H. COPPER RISK ANALYSIS
- I. NITRATE/NITRITE RISK ANALYSIS
- J. NONCANCER SPREADSHEET FOR ARSENIC
- K. CANCER SPREADSHEET FOR ARSENIC
- L. TOXICOLOGY OF METALS AND OTHER INORGANICS
- M. PARTICULATE EMISSION ESTIMATION
- N. TABLES OF DATA USED IN THE ECOLOGICAL EVALUATION
- O. ECOLOGICAL TOXICITY SCREENING ASSESSMENT METHOD
- P. EFFECT OF BACKGROUND GROUND WATER CONCENTRATIONS

ABBREVIATIONS OF CHEMICAL AND QUANTITATIVE TERMS

ABS---absorption factor through skin	F-----fluoride
ac-----acre	Fe----iron
ADI---average daily intake over a person's lifetime	FI----fraction ingested
AF----soil to skin adherence factor	ft----feet
Ag----silver	g-----gram
Al----aluminum	gal---gallons
α -----dispersivity	h-----hydraulic head
As----arsenic	ha----hectare
AsO ₄ --arsenate	HCl---hydrogen chloride
AsO ₃ --arsenite	HCN---hydrogen cyanide
AT----averaging time	HCO ₃ --bicarbonate
Au----gold	Hg----mercury
AVG---average	HQ----hazard quotient
Ba----barium	in----inch
Be----beryllium	K-----hydraulic conductivity
BNA---base, neutral and acid extractable compounds	K-----potassium
BW----body weight	kg----kilogram
C-----concentration	λ -----degradation coefficient
Co----initial concentration	lb----pound
Ca----calcium	IR----ingestion rate
Cd----cadmium	L-----hydraulic length between two wells
CF----conversion factor	L-----liter
Cl----chloride ion	LC50--median lethal concentration
cm----centimeter	LD50--median lethal dose
CN----cyanide	LDlo--lowest lethal dose
Co----cobalt	LOAEL-lowest observed adverse effect level
CO ₃ ---carbonate	m-----meter
Cr----chromium	MCL---maximum contaminant limit
Cu----copper	MDL---
CS----chemical concentration in soil	mg----milligram
CW----chemical concentration in water	Mg----magnesium
d-----day	μ g----microgram
°C----degrees Celsius	μ m----micrometer
°F----degrees Fahrenheit	μ mho--micromho
D-----dispersion coefficient	μ S----microSiemen
E-----exposure level	Mn----manganese
EC----electrical conductivity	mL----milliliter
ED----exposure duration	mm----millimeter
EF----exposure frequency	Mo----molybdenum
Eh----oxidation-reduction potential	mv----millivolts
EPTOX-extraction procedure toxicity	n-----porosity
	N-----nitrogen
	Na----sodium
	Ni----nickel
	NO ₃ ---nitrate

NO₂---nitrite
OH----hydroxyl
Pb----lead
pe----negative logarithm of
 electron activity
pH----negative logarithm of
 hydrogen ion activity
ppb---parts per billion
ppm---parts per million
R-----retardation coefficient
RfD---reference dose
RME---reasonable maximum
 exposure
s-----second
SA----surface area available
 for contact
Sb----antimony
Se----selenium
SF----slope factor
Si----silicon
SI----saturation index

Sn----tin
SO₄---sulfate
t-----time
TCLP--toxic characteristic
 leaching procedure
TDlo--lowest toxic dose
Tl----thallium
TOC---total organic carbon
U.B.--upperbound
v-----specific discharge or
 Darcy velocity
v-----average linear
velocity or effective
 velocity
V-----vanadium
VOC---volatile organic
 compounds
WAD---weak acid dissociable
x-----distance
yd----yard
yr----year
Zn----zinc

EXECUTIVE SUMMARY

The Silver Mountain Mine site comprises about five acres in Okanogan County, Washington, which have been contaminated with mining wastes containing cyanide, arsenic, and other metals. The mine site is six miles northwest of Tonasket along the west margin of Horse Springs Coulee, a north-south trending valley. Horse Springs Coulee contains as much as 150 feet of unconsolidated glacial drift and alluvium overlying metasedimentary bedrock. Unconsolidated sediments thin toward the valley wall in the area of the mine site. The region is semi-arid with scrub vegetation and is used primarily for cattle grazing.

Underground mining for silver, gold, and copper production began at the site in 1902. Mining occurred in silicified zones of disseminated sulfides in the bedrock. By 1956, sporadic development produced about 2000 feet of underground mine workings and a few thousand tons of mine dump consisting of waste and mineralized rock. A 400-ton per day mill was constructed in 1952, but may never have been used. The mill has since been removed.

From 1980 to 1981, Precious Metals Extraction, Ltd., constructed and operated a cyanide leach heap of previously mined material in an attempt to extract silver and gold. The heap consisted of about 5300 tons of ore in a 100 X 105 X 14 foot pile on top of a 20-mil plastic liner. About 4400 pounds of sodium cyanide was mixed with water and sprayed on the top of the heap. The cyanide-laden effluent was then collected in a leachate pond at the base of the heap. The leach heap operation was abandoned in 1981 without cleanup of contaminated material.

The Washington Department of Ecology investigated the site in 1981 and in 1982 used sodium hypochlorite to neutralize the leachate pond and heap. The U.S. Environmental Protection Agency conducted a Preliminary Assessment and Site Inspection in 1984. The site was added to the National Priority List of Superfund sites in 1984. In 1985, the Department of Ecology conducted a site stabilization effort which included removal of liquids from the leachate pond and installation of a 33-mil plastic cover over the heap and pond to reduce infiltration. Empty cyanide drums were also removed, a fence was installed, and the site was posted. A Remedial Investigation and Feasibility Study under an Interagency Agreement with the U.S. Bureau of Mines was commenced by EPA in 1988.

The physical and chemical characteristics of the site and the nature and extent of contamination were evaluated by field

RI EXECUTIVE SUMMARY
PAGE 2

geologic mapping, hydrogeological investigation, and chemical and petrographic analysis of site materials. The hydrogeological investigation incorporated four monitoring wells and three offsite water supply wells, and two surface water sites. Thirty-four samples of leach heap and mine dump material, twenty samples of nearby soils, and three rounds of water samples from the seven wells and two surface water sites were collected and analyzed. The investigation identified and evaluated four potential sources of contaminants: the leach heap, mine dump, mine drainage, and bedrock. Potential exposure pathways for contaminants were identified as onsite soils, onsite surface water, onsite ground water in a shallow aquifer, and offsite ground water in the Horse Springs Coulee-Aeneas Lake aquifer.

Elevated levels of contaminants in solid material are largely confined to mined bedrock that has been crushed through the process of mining. The mined material has been either abandoned in unleached piles (mine dump), or abandoned after leaching with cyanide solutions (leach heap). Relative to background soils, levels of arsenic, antimony, lead, and other metals and metalloids are elevated in the mine dump, and these same constituents plus cyanide are elevated in the leach heap. The same contaminants occur at lower, but still elevated, concentrations in shallow soils beneath the heap leachate collection pond and in a localized area of shallow soil within 25 feet adjacent to the heap.

Onsite concentrations of ground water contaminants were compared to concentrations in downgradient water supply wells in the Horse Springs Coulee aquifer. Contaminants which originated at the leach heap and which were elevated in onsite ground water included cyanide, and slightly elevated levels of sodium, potassium, nitrate, nitrite and fluoride. Elevated contaminants which originated either at bedrock or at the mine dump included arsenic, antimony, barium, chromium, copper, chloride, iron, lead, manganese, nickel, silver, and zinc. In addition, elevated arsenic and antimony occur in mine drainage and originate in the mine workings in bedrock.

Ground water contaminants from the leach heap extend in a plume at least as far downgradient as the furthest monitoring well, Well 3, 50 feet southeast of the heap. Ground water contaminants from either the mine dump or bedrock are substantially reduced at Well 3, which is 100-200 feet downgradient of these potential sources. No ground water contaminants influence the nearest offsite water supply wells 2-4 miles downgradient to the southeast of the mine site.

The future impact of cyanide on ground and surface water is primarily controlled by the amount and form of cyanide remaining in the heap, and by seepage and degradation rates. Measurements

of total and weak acid dissociable cyanide indicate that the cyanide in heap material is mostly in the form of poorly soluble iron cyanide compounds. The estimated degradation and seepage rates for cyanide indicate that the levels now occurring in ground water probably originated in spillage or leachate pond overflow at the time of leach heap operations during 1980 and 1981. Probably little, if any, leachate has been produced since the heap was covered in 1985. However, with time and deterioration of the plastic top and bottom liners, leaching of cyanide from the heap would be expected to resume with transport of cyanide to ground water. Infiltration of cyanide is projected to occur at progressively reduced concentrations and rates as a result of degradation, including speciation to hydrogen cyanide (HCN) and subsequent volatilization. Projected maximum concentrations of cyanide in leachate are on the order of a few milligrams per liter. Infiltration of leachate at these concentrations is projected to degrade to significantly lower levels during passage through the unsaturated zone.

The future impact of arsenic on ground and surface water is primarily controlled by the amount and form of arsenic in all mined materials, including the heap and mine dump, by the amount and form of arsenic in bedrock, and by the sorption capacity of iron- and aluminum-rich soils. The estimated solubility of arsenic and sorption capacity of soils indicate that, as the surficial piles oxidize, leachate from the heap and the mine dump could produce high concentrations of arsenic on the order of a few tens of milligrams per liter. Retardation of initially high concentrations of arsenic in leachate could occur during infiltration. However, with time and saturation of sorption sites, arsenic levels impacting ground water could reach the same levels of arsenic as infiltrating leachate. Current levels of arsenic in ground water indicate that oxidation of the mine dump and buried bedrock has not yet progressed to the point of producing highly concentrated leachate. Current elevated levels of arsenic in mine drainage, however, indicate that oxidation may now be taking place in the mine workings. Consequently, a potential exists for arsenic concentrations to increase in leachate from any of the mined material continually exposed to the oxidizing influences of weathering or water infiltration.

At present, water supply wells in the main part of Horse Springs Coulee aquifer are not affected by contaminated ground water from the mine site. The projected impact from estimated future levels of contaminants is significantly less in the Horse Springs Coulee aquifer than in the shallow aquifer at the mine site because of dilution resulting from a large contrast in ground water flow between the two areas.

The human health risk from cyanide, arsenic, and other contaminants is based on the likely future use of the site.

RI EXECUTIVE SUMMARY
PAGE 4

Industrial (mining) activity is expected to be the most reasonable future use of the property, though currently there is little activity. Given this assumption, the most important exposure routes are ingestion of and dermal contact with soil, and ingestion of ground water or surface water. Exposure via inhalation of suspended particulates or volatile chemicals (HCN) is not expected to be an important exposure route.

Using reasonable maximum exposure assumptions, arsenic, antimony, and cyanide are the most important contaminants in water with respect to human health. Nitrate/nitrite and lead were each present in a single ground water sample at concentrations above established drinking water criteria, though these values may not be representative of overall site conditions. Exposure to arsenic in water could result in an increased cancer risk of 2×10^{-4} . There is also a risk of noncarcinogenic effects, mainly neurologic, liver, and skin related, from arsenic, cyanide, and other chemicals. The hazard quotient for these effects is 2.5.

The most important contaminant in soil is arsenic. Exposure to soil could result in an increased cancer risk of 2×10^{-3} . The hazard index of 2.4 indicates that exposure to soil could also result in a risk of noncarcinogenic effects, principally skin and neurologic disorders.

Major components of the assessment which decreased the certainty of the results were the toxicity reference values used, assumed future land use, dermal contact pathway risks, and water data. Due to the uncertainty in these and other areas, conservative assumptions were made in order to be protective of human health. Therefore, noncancer and cancer risk estimates must be carefully interpreted.

The environmental risk from exposure to contaminants on site is based on potential effects on wildlife and other biota in the sagebrush ecosystem. Wildlife in semi-arid areas are attracted to surface waters. If more leachate is generated in the heap and collects in the leachate pond, local acute effects on wildlife could occur. Because the pad liner has deteriorated, it is assumed that any leachate generated will enter the groundwater rather than ponding (except perhaps during heavy storms) and will not be a source of acute toxicity.

The temporary cover on the heap should prevent leachate from forming, prevent dispersal of soils from the heap, and restrict access to the heap by large organisms and colonizing vegetation. In addition to being covered, due to the large particle sizes and likely absence of normal sagebrush soil biota, the heap is not the equivalent of soil. As a result, few if any organisms utilize it as habitat. However, as the cover deteriorates over

RI EXECUTIVE SUMMARY
PAGE 5

time, vegetation, wildlife, and other biota could potentially be exposed to toxic concentrations of metals in ponded heap leachate or in heap soils. Soils in the heap and dump are most likely to be toxic if they erode, spread out, leach, or are otherwise made more available to onsite biota.

Air transport of particulates from the tailings pile is negligible under present conditions. Ground water is not toxic to plants or aquatic biota at present. Surface water transport is absent for most of the year and the intermittent streams do not feed the closest surface water bodies of concern. Transport to these nearby sensitive communities in Horse Springs Coulee does not occur by either surface water or ground water discharge from the site.

Although small in area, the soils nearby the heap and dump are contaminated with arsenic, manganese, selenium, and zinc, at concentrations that can affect vegetation and animals. In particular, ruminants, rabbits, rodents, and birds are at risk when consuming vegetation, soil biota, and associated soil from these contaminated soils. Manganese and selenium concentrations are of concern throughout the site, including background areas.

LIST OF FIGURES

	PAGE
1.1 Site location map.....	3
1.2 Detailed site map.....	4
2.1 Site topographic map.....	2
2.2 Site geologic map.....	3
2.3a Rock and soil sample locations.....	11
2.3b Rock and soil locations, including background.....	12
2.4 Surface water bodies.....	20
2.5 Well locations.....	22
2.6 Well diagram - Well 1.....	25
2.7 Well diagram - Well 2.....	26
2.8 Well diagram - Well 3.....	28
2.9 Well diagram - Well 4.....	27
3.1 Cross sections of mine site.....	10
3.2 Monitoring well static water levels.....	12
3.3 Ground water surface - April 1989.....	16
3.4 Ground water surface - May 1989.....	17
3.5 Ground water surface - June 1989.....	18
3.6 Ground water surface - July 1989.....	19
3.7 Monitoring well temperatures.....	20
4.1 Distribution of cyanide in the heap, dump, and soil.....	3
4.2 Distribution of arsenic in the heap, dump, and soil.....	4
4.3 Piper diagram of major cations and anions.....	24
4.4 Conductivity, cyanide, arsenic, and nitrate in water.....	25
4.5 Correlation between total and dissociable cyanide.....	29
5.1. Conceptual model of the hydrogeologic system.....	3
5.2. Cyanide concentration in leachate (1981 to 1984).....	20
6.1 Conceptual pathway model.....	16
7.1 Conceptual model of food web connections.....	24

LIST OF TABLES

	PAGE
1.1 Investigations During 1981 to 1984.....	10
2.1 Cyanide compounds and complexes.....	5
2.2 Inorganic analytical parameters.....	6
2.3 Organic analytical parameters.....	7
2.4 EP toxicity criteria.....	9
2.5 Locations of leach heap and mine dump samples.....	13
2.6 Locations of soil samples for metals and CN.....	16
2.7 Locations of soil samples for organics.....	17
2.8 Summary of procedures for water samples.....	30
3.1 Inorganic rock/soil sampling field data.....	4
3.2 Particle size distribution.....	7
3.3 Results for field parameters.....	13
4.1 Leach heap, mine dump, and soils inorganic composition.....	5
4.2 Leaching characteristics of heap material.....	7
4.3a Soils composition. Volatile organic compounds.....	10
4.3b Base neutral and acid extractable organic compounds.....	11
4.4 Water composition, Round 1. Metals and metalloids.....	17
4.5 Water composition, Round 1. Anions and cyanide.....	19
4.6 Water composition, Round 2. Anions and cyanide.....	20
4.7 Water composition, Round 3. Metals and metalloids.....	21
4.8 Water composition, Round 3. Anions and cyanide.....	23
4.9 Cyanide and arsenic concentrations and masses.....	27
5.1 Species distribution for cyanide.....	12
5.2 Saturation indices for cyanide.....	13
5.3 Species distribution for arsenic.....	16
5.4 Saturation indices for arsenic.....	17
6.1 Noncarcinogenic effects and reference values.....	2
6.2 Contaminant concentrations used to estimate risk.....	19
6.3 Carcinogen classification and potency.....	9
6.4 Arsenic carcinogenic risk.....	27
6.5 Reasonable maximum exposure (RME) risks by media.....	30
6.6 Noncarcinogenic risk by pathway.....	29
6.7 Noncarcinogenic risk by chemical.....	31
6.8 Stock tank drinking water risks (RME).....	32
6.9 Noncarcinogenic risk grouped by critical effects.....	35
6.10 Uncertainty assessment.....	36
6.11 Drinking water regulations and guidance.....	37
7.1 Species of concern at mine site.....	21

CHAPTER 1 - INTRODUCTION

1.1 PURPOSE AND SCOPE OF REPORT

The Silver Mountain Mine in Okanogan County, Washington, was included on CERCLA's (Comprehensive Environmental Response, Compensation, and Liability Act of 1980, "Superfund") National Priorities List (NPL) in October, 1984, upon the initiation of the Washington Department of Ecology (Ecology). Investigations of the site indicated that concentrations of cyanide from secondary mining presented a potential environmental and public health hazard, and cyanide contamination of ground water in the area was considered possible.

As part of the Superfund process, a Remedial Investigation and Feasibility Study (RI/FS) was initiated in September, 1988. The purpose of the RI is to gather sufficient data to identify the nature and extent of chemical contamination and to evaluate potential risks to public health and the environment. The Feasibility Study identifies and assesses remedial action alternatives, based on results of the Remedial Investigation. Completion of the RI/FS culminates in public comment on the alternatives considered and selection of a remedial alternative which mitigates threats to and provides protection of human health and the environment.

The Remedial Investigation for Silver Mountain Mine is based on a Bureau of Mines (Bureau) investigation and report completed under Interagency Agreement Number DW14933475-01-0 (dated September 1, 1988) between the U.S. Environmental Protection Agency (EPA) and the Bureau. The investigation and report were conducted by the Bureau's Western Field Operations Center (WFOC), based in Spokane, Washington, and Albany Research Center (ARC), Albany, Oregon.

The primary responsibilities of the Bureau's Western Field Operations Center was to describe site hydrology and geology, to collect representative soil and water samples, and to determine volumes of hazardous solids and liquids at the site. The Albany Research Center was responsible for conducting laboratory analysis of most of the soil and water samples. Characterization of the nature and extent of contamination at the site was completed by EPA on the basis of the Bureau's investigation. EPA supplemented the Bureau's investigation with site visits and research to complete the assessments of environmental and human health risks for the RI report.

Based on information about the cyanide heap leaching operation at the site, an abandoned leach heap and adjacent mine dump are considered to be sources of contamination. The Remedial Investigation addresses potential contamination in the leach heap, the mine dump, onsite soils and surface water, and underlying ground water. Potential releases of hazardous materials to the air are considered in the risk assessment portion of the RI report.

1.2 SITE BACKGROUND

This section includes a brief description of the site, information about historical development and land use practices, a summary of events leading to the site's inclusion on the NPL, and a discussion of the results of previous investigations.

1.2.1 Site Description

The Silver Mountain Mine site consists of five acres in Okanogan County, north-central Washington (southwest quarter of Section 34, T38N, R26E). The site (Figure 1.1) is six air miles northwest of the town of Tonasket (population 1055) and lies in a north-south trending basin between a scarp on the west and a low ridge on the east. The valley is part of a larger north-south running valley known as Horse Springs Coulee.

The area surrounding the site is semi-arid with scrub vegetation and is used primarily for cattle grazing. From county road 9410, an unpaved access road leads 1.5 miles to the site, which is surrounded by a barbed-wire fence.

Of key interest at the site is a heap of mined material and a trench remaining from an abandoned cyanide heap leaching operation (Figure 1.2). These will be referred to as the leach heap and leachate pond in this report. Both the heap and the pond are presently covered with a scrim-reinforced Hypalon liner, to be referred to as the cover. Directly west of the leach heap is a larger pile of unprocessed mined material, which will be referred to as the mine dump.

The foundations of a former mill building are about 250 feet southwest of the heap. A mine entrance, or adit portal, is located approximately 200 feet west of the heap in the scarp, and water from saturated mine workings is piped from within the portal to a cattle watering trough, or stock tank, outside the fenced area. Approximately 75 feet south of the heap was a shallow well, now sealed and abandoned. A small freshwater seep northwest of the heap creates a small shallow pool of standing water. A single tree provides shade and seasonal greenery at the site.

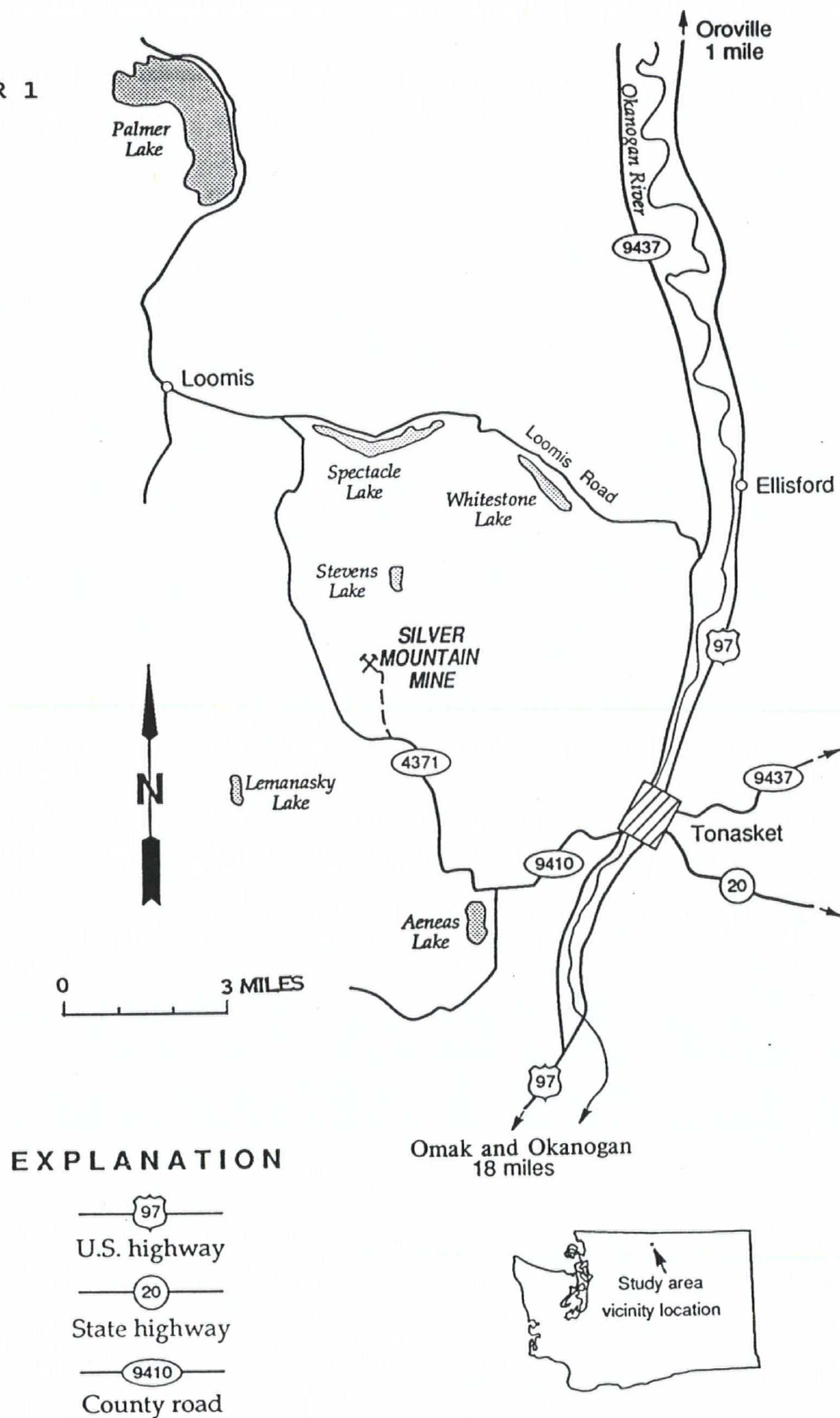


Figure 1.1. Site location map. Silver Mountain Mine is located in north central Washington, about 6 miles northwest of Tonasket.

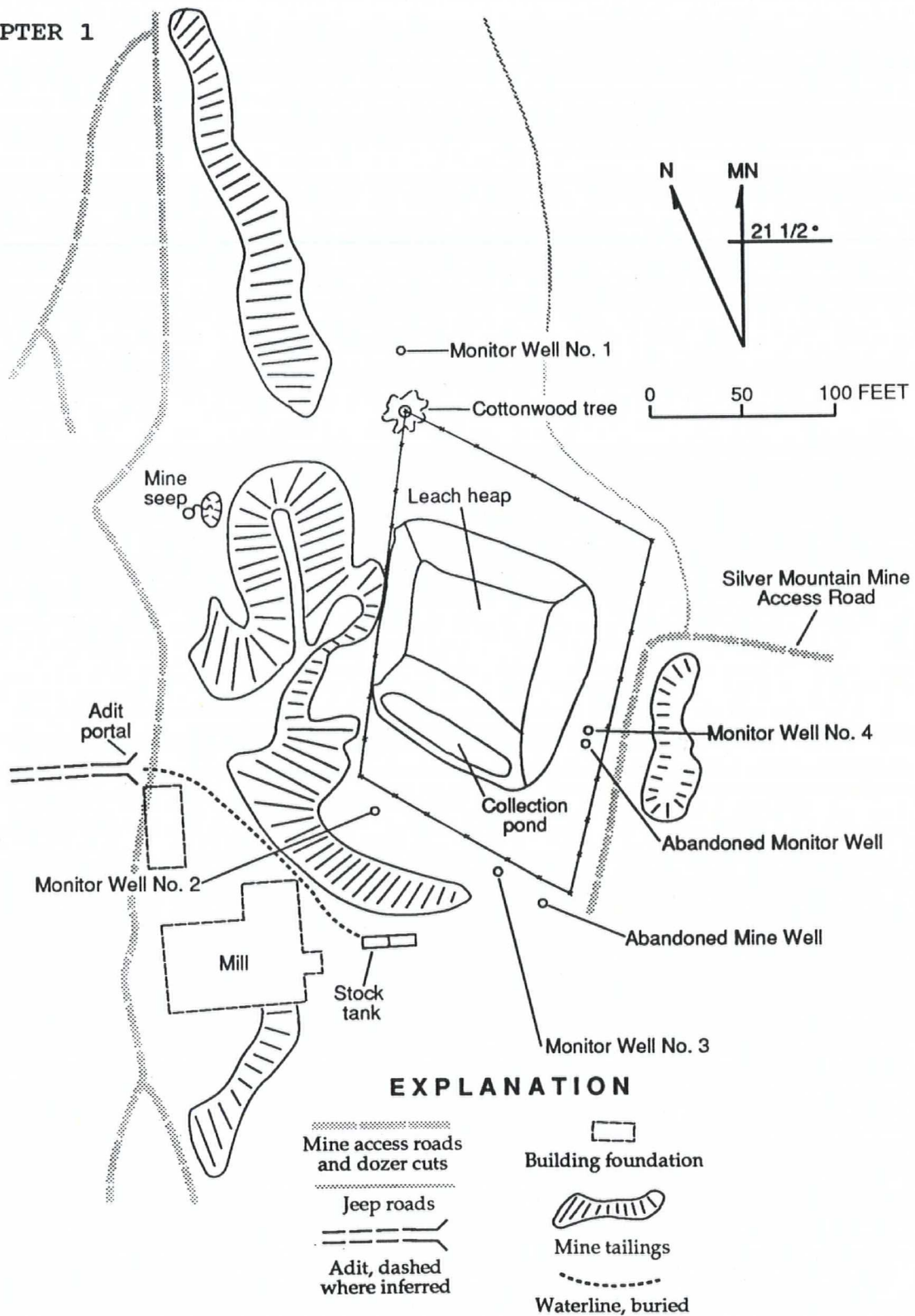


Figure 1.2. Detailed features of the Silver Mountain Mine site.

1.2.2 Site History and Land Use

Bureau of Mines records show that the Silver Mountain Mine was originally developed as the Silver Star in 1902 by the Silver-Star Mining Corp., Tonasket, Washington. Huntting (1957) reports that Lucky Lady, Inc., operated the mine in 1936. Bureau records also show that 51 tons of crude ore containing one troy ounce of gold, 63 troy ounces of silver, and 144 pounds of copper were mined at the site and smelted by the ASARCO smelter, Tacoma, Washington, in 1943. One ton of crude ore containing 2 ounces of silver was also mined there and smelted by the Tacoma smelter in 1945. No other records of production exist.

The Silver Mountain Mining Company, Inc., of Tacoma, Washington, acquired the mineral rights to the property in 1951. In 1952, a 400-ton-per-day capacity mill was constructed on the site (Huntting, 1957). By 1956, sporadically developed underground mine workings totalled 2,000 feet (Huntting, 1957). Aerial photographs of the site (appendix B) reveal that the mill building and equipment were dismantled prior to August 1977. Observations made by Bureau personnel while collecting field data for the Remedial Investigation indicate that little, if any, ore was ever processed at the mill. The large heap of mined material used in the cyanide leaching process is composed of ore and rock broken in the course of mining. According to Huntting (1957), mining took place in silicified zones 1-20 feet wide which contained disseminated sulfide mineralization with assay values in silver, gold, lead, and zinc. Bureau records show only silver, gold, and copper production.

In 1979, the Silver Mountain Mining Company, Inc., changed its name to Lead Point Consolidated Mines Company (Woodward-Clyde Consultants, 1987). In February of 1980, Lead Point Consolidated Mines leased the mineral rights to a limited partnership of J. Wayne Tatman and G. Patrick Morris of J.W.T. and G.P.M., Ltd. (Woodward-Clyde Consultants, 1987). Morris and Tatman operated the property as Precious Metals Extractions, Ltd. (PME) (Ecology and Environment, 1985). In 1985, both Lead Point Consolidated Mines and PME were owned in partnership by J. Wayne Tatman and G. Patrick Morris (Ecology and Environment, 1985).

From late 1980 to late summer 1981, Precious Metals Extraction, LTD attempted to extract silver and gold from mine dump material using cyanide heap leaching techniques (Ecology and Environment, 1985). Typical cyanide heap leaching practice is to pile ore on a graded impervious liner system and sprinkle the ore heap with an alkaline cyanide solution containing little or no silver and gold. The solution percolates through the heap, chemically bonding with metals in the ore, and the metals-containing solution gathers in a collection pond. Precious

metals are then recovered from the solution. The used solution is typically adjusted for pH and cyanide content and is re-applied to the top of the ore heap until sufficient recovery of the precious metals is made. Additional information on cyanide heap leaching techniques is provided in Appendix A.

No historical records or company staff are available to describe the process used during the development by Precious Metals Extraction (PME). The following scenario is based on field observations and data collected by the Bureau in 1989. To begin the heap leaching process, PME cleared an area approximately 180 feet by 140 feet, adjacent to existing mine dumps. A leach pad base of sandy soil, up to 3 feet thick and graded with a 2.5 percent slope to the southwest, was then prepared. At the southern end of the sloped pad base a rectangular trench 7 feet by 75 feet and averaging 4 feet in depth, was dug as a leachate collection pond. The soil base and pond were then covered with a green 20-mil thick plastic liner. Another layer of sandy soil, from 0 to 6 inches thick, was then placed over the plastic liner. Last, approximately 5,300 tons of material from the mine dump were loaded onto the pad. The prepared heap was approximately 100 feet long, 105 feet wide, and 14 feet high.

Ecology and Environment (1985) report that during the months PME operated the heap leach, several tons of caustic soda and lime and approximately 4,400 pounds (20 55-gallon drums) of granular sodium cyanide were combined with water and pumped over the heaped material on the pad. After the alkaline cyanide solution percolated through the heap and drained into the collection pond, the remaining processing sequence is unclear. Ecology and Environment (1985) state that gold and silver were electroplated directly from the metals-laden leachate and that the alkalinity and cyanide content of the leachate were adjusted before reapplication of the solution to the heap. Woodward-Clyde (1987) report that activated carbon was used to remove the silver and gold from the leachate. Direct electrowinning of leach solutions is possible, but the two most commonly used methods for removing gold and silver from alkaline cyanide heap leach solutions are Merrill-Crowe zinc dust precipitation and activated carbon adsorption. Photographs taken by the Department of Ecology in July 1981 and July 1982 (Appendix B) suggest that solution was pumped from the leachate pond into barrels of activated carbon lined up next to the pond. Excess solution was allowed to overflow the barrels onto a plastic liner and run back into the pond. The photos also indicate that the carbon, containing gold and silver, may have been pressure-stripped of the precious metals on site.

Available information does not indicate whether an additional tank or pond was used to adjust the alkalinity and

cyanide content of the leachate before pumping it back on top of the heap. It is possible that the operators used the single collection pond for all processing steps. The photographs cited above show two small mixers which may have been used for alkalinity adjustment. A solution distribution pump is also shown.

Water used for the leaching process appears to have been obtained from inundated mine workings. A 1981 photograph shows a black PVC pipe leading from inside the abandoned mine to the leach operations area. Water from within the mine is now used to fill a stock watering trough. The photos also show an abandoned oil tank truck. This may have been used to deliver water or to store fuel for the equipment used during heap construction.

1.2.3 Regulatory History

The following sequence of events leading to the listing of the Silver Mountain Mine site on the National Priorities List is based on a 1985 report by Ecology and Environment (E & E), the community relations plan prepared by Woodward-Clyde Associates (1987), and material in the EPA site file. Complete data from sampling events are shown in Table 1.1.

In June 1981, the owner of the surface rights to the property informed Okanogan County Health Department officials of the heap leaching operation. The Health Department collected samples from two of the barrels lined up by the leachate pond, the pond, the heap, and the area between heap and pond. The Washington Department of Ecology lab analyzed the samples for metals in the liquids and for total cyanide in all. Total cyanide results were 2000 mg/l and 1500 mg/l in the two barrels, 1100 mg/l in the pond, 390 mg/kg in the heap, and 360 mg/kg in the area between. Signs reading "POISON - KEEP OUT" were posted.

After late July 1981, PME abandoned the site without neutralizing the solution in the leachate pond or the materials on the leach pad. Empty cyanide drums and several large containers of carbon also remained on site.

In October of 1981, the Department of Ecology took samples from the shallow well 75 feet from the leach heap and analyzed for metals and total cyanide. Results showed <0.003 mg/l total cyanide in the well and 430 mg/kg in the heap.

On November 13, 1981, the Superior Court of the State of Washington issued an order for Okanogan County officials to enter the site and address potential health hazards posed by the leachate in the collection pond and by discarded chemical and processing containers.

The owner of the property surface rights put temporary fencing around the site to prevent cattle from exposure to contaminated pond liquids in 1981.

In November 1981, Department of Ecology sampling indicated total cyanide concentrations of 600 mg/l in the leachate pond and <0.002 mg/l in the onsite well. Ecology neutralized leachate in the collection pond with sodium hypochlorite (HTH) which converts cyanide to carbon and nitrogen. Free chlorine was observed, indicating that neutralization was complete. Using a water tanker-spray truck, Ecology sprayed the neutralized solution around the mine area. Additional HTH was put into the trench to neutralize any leachate that might collect over the winter.

In Spring of 1982, Department of Ecology sampled the winter leachate and found total cyanide values of 220 mg/l. Soil where the neutralized liquids had been disposed of in November 1981 were also sampled and indicated 0.22 mg/kg total cyanide. HTH solids which had settled in the pond were stirred up to activate neutralization.

In December 1982, the pond was again neutralized by the Department of Ecology. Reportedly, the liquid was circulated repeatedly through the heap during neutralization. Over 5 hours, total cyanide levels in the pond dropped from 19 mg/l to <0.007 mg/l. Small amounts of liquid coming from the heap after the process were measured at 30 mg/l total cyanide. A soil sample taken near the pond showed 100 mg/kg total cyanide. In November 1983, samples from the leachate pond indicated that leaching by rain and snowmelt through the heap had brought total cyanide concentrations in the pond up to 9.2 mg/l.

In September of 1984, Ecology and Environment, Inc. (E & E) conducted a preliminary site inspection for EPA and made recommendations for neutralizing cyanide in the leach heap. Leachate collection pond liquids, heap soil, and onsite background soil were analyzed for total cyanide and metals. E & E collected two water and two soil samples for the Department of Ecology.

Results ranged from 50 mg/kg of total cyanide one foot from the heap to 480 mg/kg in the heap. Samples of liquids in the pond showed 2.3 mg/l and 7.6 mg/l total cyanide. The onsite well showed <0.001 mg/l.

The Department of Ecology proposed the site for inclusion on the NPL in 1981, 1982, and 1983. In October 1984 the site was included on the NPL.

The Department Ecology samples from November 1984 showed concentrations of total cyanide at 2.3 mg/l in the pond. A white crystalline substance found on the underliner between the heap and the trench showed 2400 mg/kg total cyanide.

In June 1985, the Department of Ecology conducted a site stabilization procedure. Liquids in the collection pond were pumped out, and all residue was removed. Wastes were taken to a treatment facility in Kent, Washington. The heap and pond were covered with a geotextile fabric and a 33-mil hypalon liner to prevent rain and snowmelt from leaching through the heap and accumulating in the pond. All empty cyanide drums were removed. The barbed wire fence, removed during stabilization procedures, was reinstalled.

In November 1988, at the request of EPA officials, Bureau of Mines personnel permanently abandoned the shallow well 75 feet south of the leach heap. Although the 20-foot well showed no apparent cyanide contamination during earlier investigations, it had the potential to act as a conduit for contaminants to enter the aquifer. The well was sealed by filling the well casing with granular bentonite to within 4 feet of the ground surface and filling the remaining void with a bentonite-concrete mix.

Table 1.1 Chemical Results from Previous Site Investigations
During 1981 to 1984

JUNE 1981

	Barrel #5 mg/l	Barrel #12 mg/l	Leachate Pond mg/l	Top of Leach Heap mg/kg	Between Heap and Pond mg/kg
Total CN	2000	1500	1100	390	360
Cu	1.5	4.5	100.	-	-
Zn	190.	140.	92.	-	-
Fe	250.	340.	110.	-	-
Ni	0.45	0.45	3.0	-	-
Cr	0.04	0.17	0.04	-	-
Cd	1.08	0.40	0.82	-	-
Pb	0.16	0.16	0.22	-	-
Ag	0.70	0.12	0.91	-	-

OCTOBER 1981

	Well mg/l	Leach Pond mg/l
Total CN:	<0.003	430.
Cu	<0.05	0.92
Zn	0.42	11.2
Ni	<0.05	2.38
Cr	<0.02	<0.02
Cd	<0.02	0.22
Pb	<0.2	<0.2

NOVEMBER 1981

	Well mg/l	Pond (before treat) mg/l	Pond (after treat) mg/l
Total CN:	<0.002	600.0	^a 520.
Metals: (unfiltered/filtered)			
Cu	-	-	11.0/0.09
Zn	-	-	50.0/4.8
Fe	-	-	9.3/0.23
Cd	-	-	0.14/0.10
Pb	-	-	0.25/0.20
Hg	-	-	.042

^aThis value is believed to represent sampling or analytical error

April 1982

	leachate mg/l	leachate mg/l	soil mg/kg
Total CN:	220	220	0.22

DECEMBER 1982

Progressive samples of leach pond during leachate neutralization. Sample 11 is runoff from heap after treatment.

Sample #:	1 mg/l	2 mg/l	3 mg/l	4 mg/l	11 mg/l	12 mg/l
Total CN:	30	100	19	<0.001	9.1	<0.007

JUNE 1983

	Leach Pond mg/l	Leach Pond mg/l
Total CN:	3.2	2.6

NOVEMBER 1983

	Leach Pond mg/l
Total CN	9.2

SEPTEMBER 1984

	Leach Heap mg/kg	Base of Heap mg/kg	Leachate mg/l	Onsite Well mg/l
Total CN:	480	50	7.6	<0.001

NOVEMBER 1984

	^b Leach Heap mg/kg	Leachate Pond mg/l
Total CN:	2400	2.3
WAD CN:	120	0.7
^b white powder found between heap and pond		

SEPTEMBER 1984

	Leachate Pond Water Sample mg/l	Background Soil Sample mg/kg	Heap Soil Sample mg/kg
Cyanide	110.	<0.37	20.
Arsenic	111.	110.5	35.5
Chromium	7.	13.5	0.95
Cobalt	2500.	14.5	3.2
Copper	1670.	630.	130.
Iron	18800.	37700.	5450.
Lead	<1.7	40.5	35
Manganese	1350.	820.	168.5
Mercury	4.	<0.09	<0.09
Nickel	212.	25.	4.55
Zinc	315.	161.	100.

1.2.4 Previous Investigations

As discussed above, chemical data were collected on several occasions during earlier site investigations. Results of the sampling described above are tabulated in Table 1.1.

Although variability in sample collection and analysis procedures from 1981 to 1984 limit data comparability, the data are sufficient to indicate the presence of cyanide at the site, both in the heap material and in any leachate passing through the heap material. The investigations, sampling, and stabilization conducted between 1981 and 1985 helped focus the scope of the Remedial Investigation. Department of Ecology analyses showed that cyanide and some metals concentrations in the heap were considerably higher than background, and indicated a need for

better defining the metals content in the leach heap. Department of Ecology photographs showing a fuel tank truck on the site gave rise to concern about potential contamination from fuel spillage and suggested a need for screening of organics in a few localized areas. Observations that the liner beneath the heap and leachate pond was deteriorating indicated that ground water and soil from under the pond and near the heap should be sampled to determine whether contamination had entered surrounding soils and ground water.

1.3 REPORT ORGANIZATION

Report organization follows the suggested Remedial Investigation report format established in the EPA document "Guidance for Conducting Remedial Investigation and Feasibility Studies under CERCLA," Interim Final, 1988, Table 3-13. Briefly, the contents of the report are as follows:

Executive Summary

Chapter 1: introduces and describes the site and provides background on previous investigations at the site.

Chapter 2: outlines field work conducted to determine the physical and chemical characteristics of the site and describes EPA site visits to assess the potential for human and biota exposure to hazardous materials.

Chapter 3: presents results of field investigations of the physical characteristics of the site area.

Chapter 4: discusses the nature and extent of contamination at the site.

Chapter 5: examines contaminant fate and transport.

Chapter 6: assesses the risks to human health from exposure to contaminants at the site.

Chapter 7: assesses the risks to biota from exposure to contaminants at the site.

Appendices: contains supporting documentation for heap leach methods, field investigations, laboratory analyses and risk analysis.

REFERENCES

- Ecology and Environment, Inc., 1985. Preliminary site inspection report of Silver Mountain Mine, Loomis, Washington: CH2M Hill, Bellevue, Washington, 64 p.
- Huntting, M.T., 1956. Inventory of Washington minerals-Part 2. Metallic minerals: Washington Division of Mines and Geology, Bulletin 37, 428 p., 27 plates.
- Woodward-Clyde Consultants, 1987. Community relations plan for the Silver Mountain Mine Superfund site, Okanogan County, Washington: Woodward-Clyde Consultants, San Francisco, California, 16 p.

CHAPTER 2 - STUDY AREA INVESTIGATION

This chapter describes activities conducted during the Remedial Investigation to obtain physical and chemical information about the site. Results of these investigations are provided in Chapters 3 and 4 and are discussed in Chapter 5.

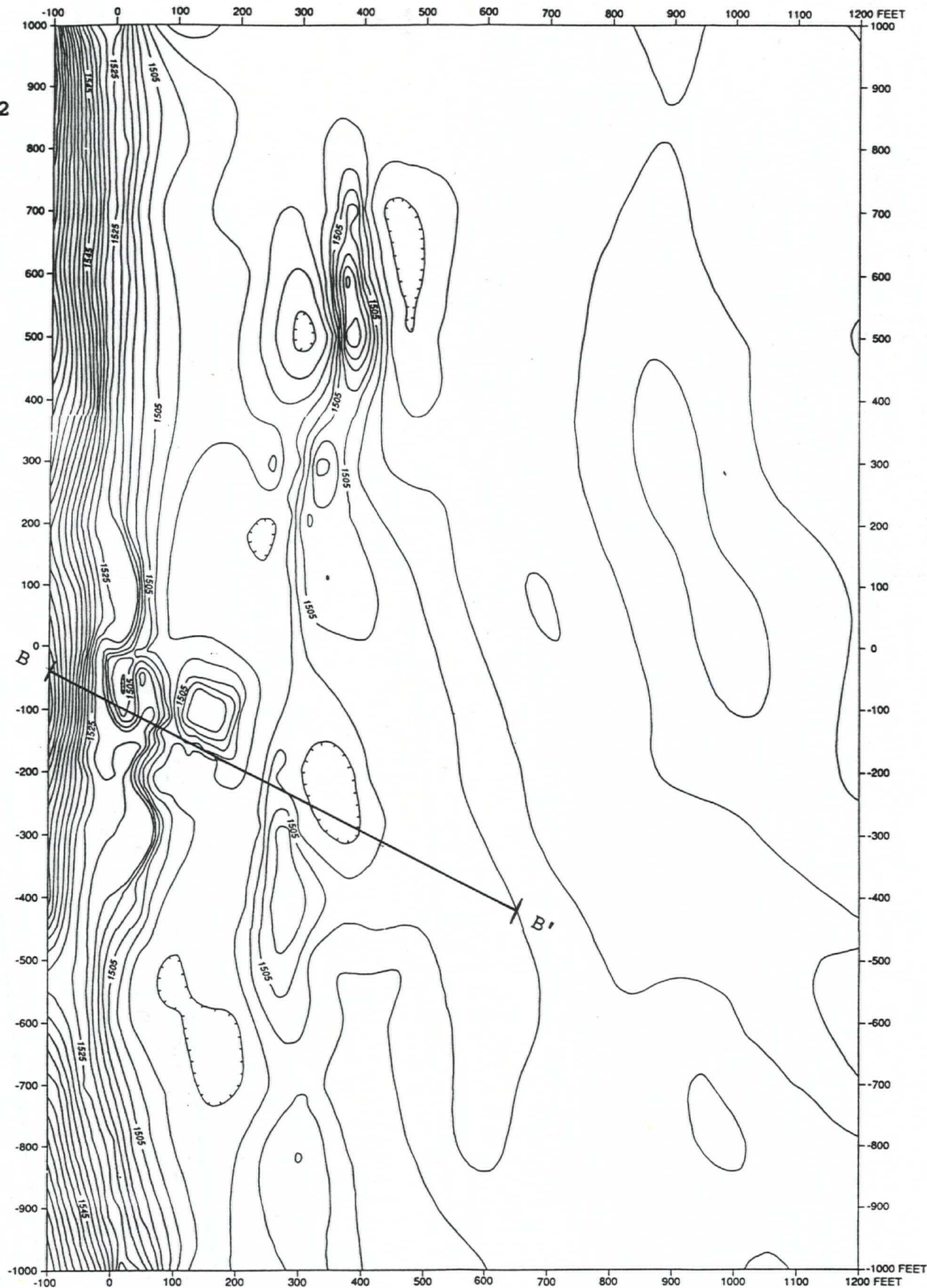
2.1 SURFACE FEATURES

A land survey was performed by the Bureau of Mines in May, 1989 to develop an accurate map of the site and determine elevations of site features. The survey area, 2000 feet northerly by 1300 feet easterly, included the surface expressions of major underground mine workings and the area's most diverse geology. After laying out the initial boundary lines, numerous north-south and east-west baselines were laid out to facilitate the survey. A total of 471 points were surveyed with a theodolite and a laser ranging device. Accuracy was 3 seconds for the theodolite vertical circle, and ± 5 mm (\pm ppm) for the laser ranging device. The finished survey was then tied into a National Geodetic Survey (NGS) benchmark off site (SWNENW, Sec 14, T37N) to obtain elevations relative to datum. Elevations in the mapped area range from 1492 to 1595 feet above mean sea level.

All survey points included elevations, X-distances, and Y-distances from a central map point, designated 0,0. The information was entered into a computer with a topographic mapping program. The resulting computer-generated map was hand adjusted based on an EPA aerial photograph of the site (Number PIC 98726, EMSL-LV, Frame no. 85746, 7/2/85) and field information to obtain a best-fit match to the area's actual physical features. The completed map is shown in Figure 2.1.

2.2 GEOLOGICAL INVESTIGATION

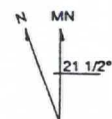
To provide information about the underlying soils and rock, Bureau of Mines personnel conducted a field geological investigation and prepared a site geologic map (Figure 2.2) for the same area included in the topographic survey. The EPA aerial photograph cited above was used for a reference base. The scale of the aerial photo was determined by comparing reference lines shown on the photo with the same lines on the ground. The scales of the original topographic and geologic maps are the same as the EPA aerial photograph used for reference (1 inch:106.4 feet).



Field Surveyed: May, 1989
by Thomas M. Sweeney, David K. Denton,
and John R. Benham



0 50 100 200 300 400 500 FEET
Contour Interval 4 Feet

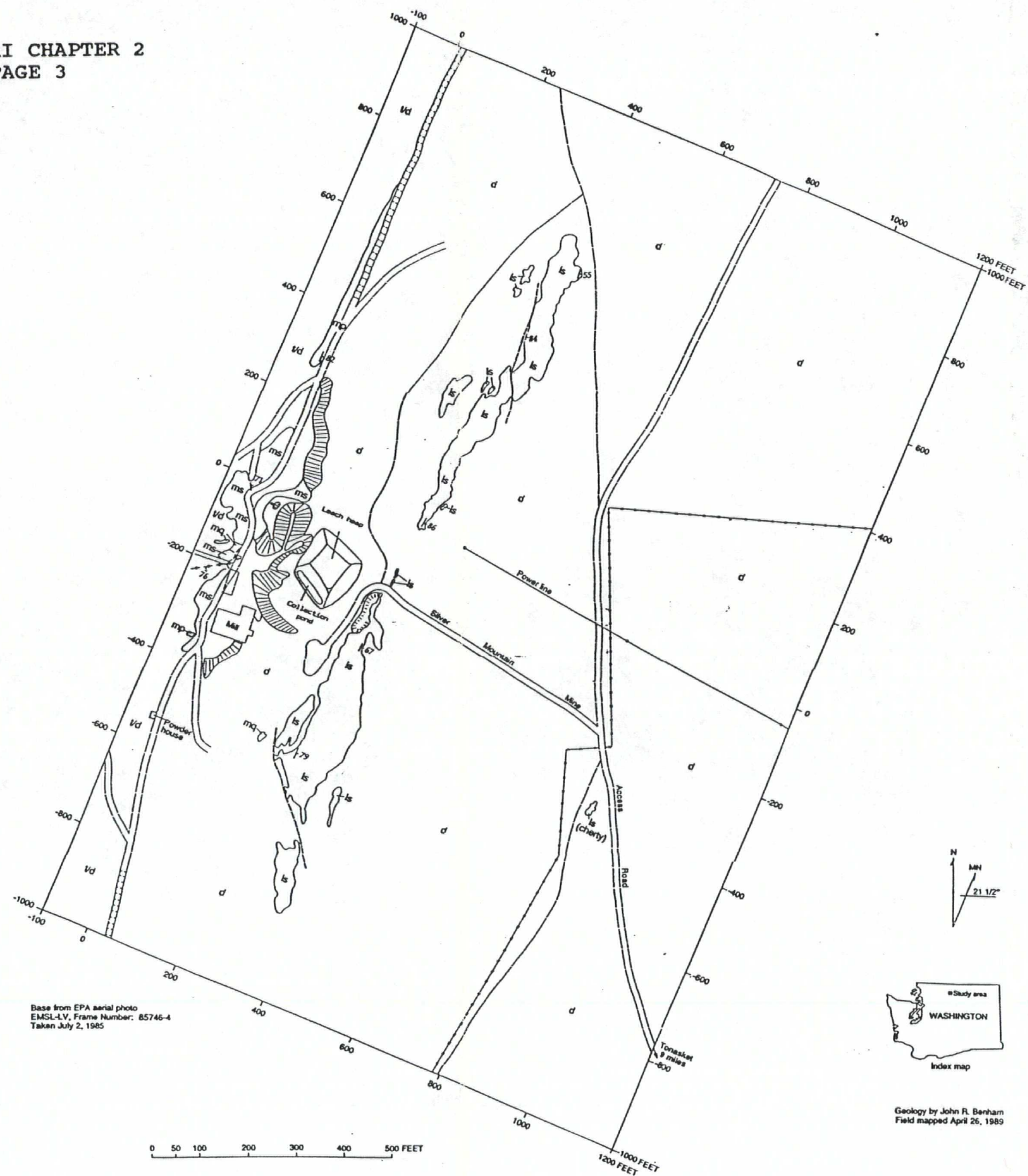


TOPOGRAPHIC MAP OF THE SILVER MOUNTAIN MINE, OKANOGAN COUNTY, WASHINGTON

By.

U.S. Bureau of Mines, Western Field Operations Center, Spokane, Washington

Figure 2.1



EXPLANATION

Mine access roads and dozer cuts

Jeep roads

d
Glacial drift
-rounded cobbles and boulders
mostly composed of granite;
strewn about an ancestral
glacial outwash plain

Vd
Talus with glacial drift
-admixture of talus debris
including metaquartzite, metaschist,
metaphyllite and glacial drift

ls
Limestone
-bedded, fine grained with
white laminations (chert), silty,
gun metal grey, and strong
HCL reaction

ms
Metaschist
-dark grey, fine grained, foliated
one direction, fracture with foliation,
weathers reddish brown; some outcrops
have chevron folding, strongly foliated
and variegated

mp
Metaphyllite
-grey to tan, weathers reddish brown,
HCL reaction in weathered planes,
foliation trends N 10 E with an 84° E
dip, very brittle

[mq]

Metaquartzite

-dark grey to ochre with white
veinlets, weathers to a buff,
slight HCL Reaction, hard,
conchoidal fracturing

77₁

Inferred vein, showing dip.

Contact

Fault, dashed where inferred

Strike and dip of bedding

Adit, dashed where inferred

 Mine building foundations

Aqueduct, abandoned

Depression with spring

Mine tailings

Fence line

GEOLOGIC MAP OF THE SILVER MOUNTAIN MINE, OKANOGAN COUNTY, WASHINGTON

By,
U.S. Bureau of Mines, Western Field Operations Center, Spokane, Washington
June, 1989

Identification of the rock types on site was performed using standard geologic field methods. Trends of outcrops and dips of strata and faults were determined with the aid of a Brunton field compass. The resulting geologic map shows all rock outcrops, their associated strikes and dips, faults and veins, and the majority of the physiographic features as seen on the ground.

2.3 ANALYTICAL PARAMETERS

On the basis of the heap leaching operations at the site and results from previous investigations, sample locations were selected for the leach heap, surrounding soils, onsite surface water, and onsite and offsite ground water.

Since the heap leaching operation used cyanide solution to extract metals from the ore, all of the water samples and most of the soil and rock samples were analyzed for metals and cyanide. When dissociated and in solution, cyanide occurs as cyanide ions (CN^-) and hydrogen cyanide (HCN) and is termed free cyanide (American Public Health Association, 1985). Prior to collection of water samples, a screening test for free cyanide was conducted in the field to determine the presence of concentrations greater than 1 mg/L.

The ratio of cyanide ions to hydrogen cyanide depends in large part on the pH of the solution. At a lower pH (i.e. in a more acidic solution), a greater proportion of HCN occurs. Some cyanide species are toxic and others are relatively inert. Table 2.1 lists typical cyanide compounds formed in a cyanide leach heap for precious metals, in order of increasing stability and decreasing toxicity.

In addition to cyanide and metals analysis, water samples were analyzed for other parameters in solution, specifically the anions listed in Table 2.2, and for alkalinity. Table 2.2 includes the target detection limits for inorganic parameters.

EPA method SM 412 was used for cyanide analysis, and ion chromatography (EPA 310.1) was used for anion analysis. Metals analyses were completed with the inductively coupled plasma atomic emission spectroscopy (ICP) and graphite furnace atomic absorption spectroscopy (GFAA).

The presence of organic contaminants at the site was investigated although historical records provided little indication of their use or disposal. Soils from a shallow pit at the north end of the site and from several other locations were sampled and analyzed for organic constituents. Volatile organic

compounds (VOC) and base, neutral and acid extractable (BNA) compounds were analyzed using gas chromatography/mass spectrometry methods, while total organic carbon (TOC) was determined using EPA method 9060. The organic analytical parameters for these soil samples are listed in Table 2.3.

Eight soil samples from the heap were also subjected to two additional tests to evaluate the leachability of metals. These tests are Toxicity Characteristic Leaching Procedure (TCLP) and Extraction Procedure Toxicity (EP-Toxicity) methods. EP-Toxicity results are used to identify hazardous waste according to the Resource Conservation and Recovery Act (RCRA) and the Washington Dangerous Waste regulations. TCLP results are used in evaluating land disposal restrictions. Metals concentrations in sample leachate that classify a substance as a RCRA hazardous waste based on EP-Toxicity tests are listed in Table 2.4.

In addition to laboratory analyses, a number of physical and chemical parameters were measured in the field. The field measurements include water temperature, pH, Eh, and electrical conductivity, and are discussed in Section 2.9.

TABLE 2.1 METAL CYANIDE COMPOUNDS AND COMPLEXES TYPICALLY FORMED IN CYANIDE LEACHING

CYANIDE SPECIES	SPECIES POSSIBLE IN HEAP
1. Free Cyanide	CN^- , HCN
2. Simple Cyanide Compounds	
a. readily soluble	NaCN , KCN , $\text{Ca}(\text{CN})_2$, $\text{Hg}(\text{CN})_2$
b. relatively insoluble	$\text{Zn}(\text{CN})_2$, CuCN , $\text{Ni}(\text{CN})_2$, AgCN
3. Weak Metal-Cyanide Complexes	$\text{Zn}(\text{CN})_4^{2-}$, $\text{Cd}(\text{CN})_3^-$, $\text{Cd}(\text{CN})_4^{2-}$
4. Moderately Strong Metal-Cyanide Complexes	$\text{Cu}(\text{CN})_2^-$, $\text{Cu}(\text{CN})_3^{2-}$, $\text{Ni}(\text{CN})_4^{2-}$, $\text{Ag}(\text{CN})_2^-$
5. Strong Metal-Cyanide Complexes	$\text{Fe}(\text{CN})_6^{3-}$, $\text{Fe}(\text{CN})_6^{4-}$, $\text{Co}(\text{CN})_6^{4-}$, $\text{Au}(\text{CN})_2^-$, $\text{Hg}(\text{CN})_4^{2-}$

Modified after Huiatt and others, 1982; and Brickell, 1981.

Table 2.2 Inorganic Analytical Parameters, and
Detection Limits

Inorganic Parameters	Detection Limits	
	Solids (mg/kg)	Water (µg/L)
<u>SOIL AND WATER SAMPLES</u>		
Cyanide, total	0.5	5.0
Cyanide, weak acid dissociable	0.5	5.0
METALS:		
Aluminum	20.0	50.0
Antimony	6.0	3.0
Arsenic	1.0	2.0
Barium	2.0	2.0
Beryllium	0.5	1.0
Cadmium	0.5	1.0
Calcium	50.0	500.0
Chromium	1.0	5.0
Cobalt	5.0	5.0
Copper	3.0	5.0
Iron	10.0	10.0
Lead	0.5	1.0
Magnesium	50.0	200.0
Manganese	2.0	2.0
Mercury	0.02	0.2
Molybdenum	1.0	10.0
Nickel	4.0	15.0
Potassium	50.0	200.0
Selenium	0.5	2.0
Silicon	50.0	200.0
Silver	1.0	5.0
Sodium	50.0	500.0
Thallium	1.0	1.0
Tin	5.0	20.0
Vanadium	5.0	10.0
Zinc	2.0	2.0
ANIONS: (Water samples only)		
Chloride	N/A	500.0
Fluoride	N/A	100.0
Sulfate	N/A	500.0
Nitrate	N/A	10.0
Nitrite	N/A	10.0
Ammonium	N/A	20.0
ALKALINITY as CaCO ₃ (Water samples only)		5000.0

Table 2.3 Organic Analytical Parameters and Range of Detection Limits (A full listing of parameters is provided in Appendix D)

Organic Parameters (Soils Only)	Target Detection Limits (mg/kg)
---------------------------------	------------------------------------

BASE NEUTRAL/ACID EXTRACTABLE ORGANICS (BNA):

	$\mu\text{g/kg}$
Phenol	330
bis(2-chloroethyl) Ether	330
2-Chlorophenol	330
1,3-Dichlorobenzene	330
1,4-Dichlorobenzene	330
Benzyl Alcohol	330
1,2-Dichlorobenzene	330
2-Methylphenol	330
bis(2-Chloroisopropyl) Ether	330
4-Methylphenol	330
N-nitroso-Di-n-Propylamine	330
Hexachloroethane	330
Nitrobenzene	330
Isophorone	330
2-Nitrophenol	330
2,4-Dimethylphenol	330
Benzoic Acid	1600
bis(2-Chloroethoxy) Methane	330
2,4-Dichlorophenol	330
1,2,4-Trichlorobenzene	330
Napthalene	330
4-Chloroaniline	330
Hexachlorobutadiene	330
4-Chloro-3-Methylphenol	330
2-Methylnaphthalene	330
Hexachlorocyclopentadiene	330
2,4,6-Trichlorophenol	330
2,4,5-Trichlorophenol	1600
2-Chloronaphthalene	330
2-Nitroaniline	1600
Dimethylphthalate	330
Acenaphthylene	330
2,6-Dinitrotoluene	330
3-Nitroaniline	1600
Acenaphthene	330
2,4-Dinitrophenol	1600
4-Nitrophenol	1600
Dibenzofuran	330

BASE NEUTRAL/ACID EXTRACTABLE ORGANICS (BNA)...continued:

2,4-Dinitrotoluene	330
Diethylphthalate	330
4-Chlorophenyl-phenyl ether	330
Fluorene	330
4-Nitroaniline	1600
4,6-Dinitro-2-Methylphenol	1600
N-Nitrosodiphenylamine(1)	330
4-Bromophenyl-phenylether	330
Hexachlorobenzene	330
Pentachlorophenol	1600
Phenanthrene	330
Anthracene	330
Di-n-Butylphthalate	330
Fluoranthene	330
Pyrene	330
Butylbenzylphthalate	330
3,3'-Dichlorobenzidine	330
Benzo(a)Anthracene	330
Chrysene	330
bis(2-Ethylhexyl) Phthalate	330
Di-n-Octyl Phthalate	330
Benzo(b) Fluoranthene	330
Benzo(k) Fluoranthene	330
Benzo(a) Pyrene	330
Indeno(1,2,30cd) Pyrene	330
Dibenz(a,h) Anthracene	330
Benzo(g,h,i) Perylene	330

TOTAL ORGANIC CARBON (TOC): 5000.0

VOLATILE ORGANICS (VOA):

	µg/kg
Chloromethane	10
Bromomethane	10
Vinyl Chloride	10
Chloroethane	10
Methylene Chloride	5
Acetone	10
Carbon Disulfate	5
1,1-Dichloroethene	5
1,1-Dichloroethane	5
1,2-Dichloroethene (total)	5
Chloroform	5
1,2-Dichloroethane	5
2-Butanone	10
1,1,1-Trichloroethane	5
Carbon Tetrachloride	5
Vinyl Acetate	10

VOLATILE ORGANICS (VOA)...continued:

	$\mu\text{g/kg}$
Bromodichloromethane	5
1,2-Dichloropropane	5
cis-1,3-Dichloropropene	5
Trichloroethene	5
Dibromochloromethane	5
1,1,2-Trichloroethane	5
Benzene	5
Trans-1,2-Dichloropropene	5
Bromoform	5
4-Methyl-2-Pentanone	10
2-Hexanone	10
Tetrachloroethene	5
1,1,2,2-Tetachloroethane	5
Toluene	5
Chlorobenzene	5
Ethylbenzene	5
Styrene	5
Total Xylenes	5

Table 2.4 Maximum Concentrations of Metals for Characteristic of EP Toxicity

	(mg/L)
Arsenic	5.0
Barium	100.0
Cadmium	1.0
Chromium	5.0
Lead	5.0
Mercury	0.2
Selenium	1.0
Silver	5.0

2.4 LEACH HEAP AND MINE DUMP INVESTIGATION

The cyanide heap leaching operation by Precious Metals Extraction, Ltd., from late 1980 to summer of 1981 is the only activity at the Silver Mountain Mine site known to have involved the use of cyanide. Historical research does not indicate that cyanide was used on the site prior to this operation. The leaching process mobilized metals in the heap, and previous investigations showed elevated levels of cyanide and metals in the leachate pond and the heap.

Prior to the Washington Department of Ecology stabilization of the site in 1985, the cyanide and metals in the leachate pond were considered the primary threat to humans and animals. Rainwater and snowmelt moved through the leach heap and refilled the collection pond seasonally. During heavy rains the pond was reported to overflow, potentially contaminating soils and ground water with cyanide compounds and metals. In addition, the possible deterioration of the plastic liner under the heap and pond may have provided another route for metals and cyanide contamination to reach ground water and soils.

Since the covering of the heap and pond with a plastic liner, the likelihood of precipitation leaching through the heap and contaminating soils and ground water has decreased considerably. The scrim-reinforced hypalon material used to cover the heap has an expected lifetime of five to ten years. However, depending on the evaluation of risk posed by metals and cyanide concentrations, more permanent remedial measures may be needed to address the heap and mine dump.

2.4.1 Leach Heap and Mine Dump Sampling Procedures

A total of 23 samples (Sample Nos. 10 - 32) of material were taken from locations on the leach heap and analyzed for metals and cyanide to characterize concentrations in the heap. The 23 samples include two duplicate samples for data quality assurance. Five samples (Sample Nos. 5 - 9) were taken of soils under the collection pond liner to investigate the possibility of liner leakage. To determine the background concentrations of cyanide and metals in the heap before leaching, three samples were taken from the mine dump (Sample Nos. 33 - 35), for a total of 31 samples.

Figure 2.3a shows the locations of all rock and soil samples. Those analyzed for cyanide and metals are designated by numbers. Table 2.5 provides brief descriptions of the leach heap and mine dump sample locations.

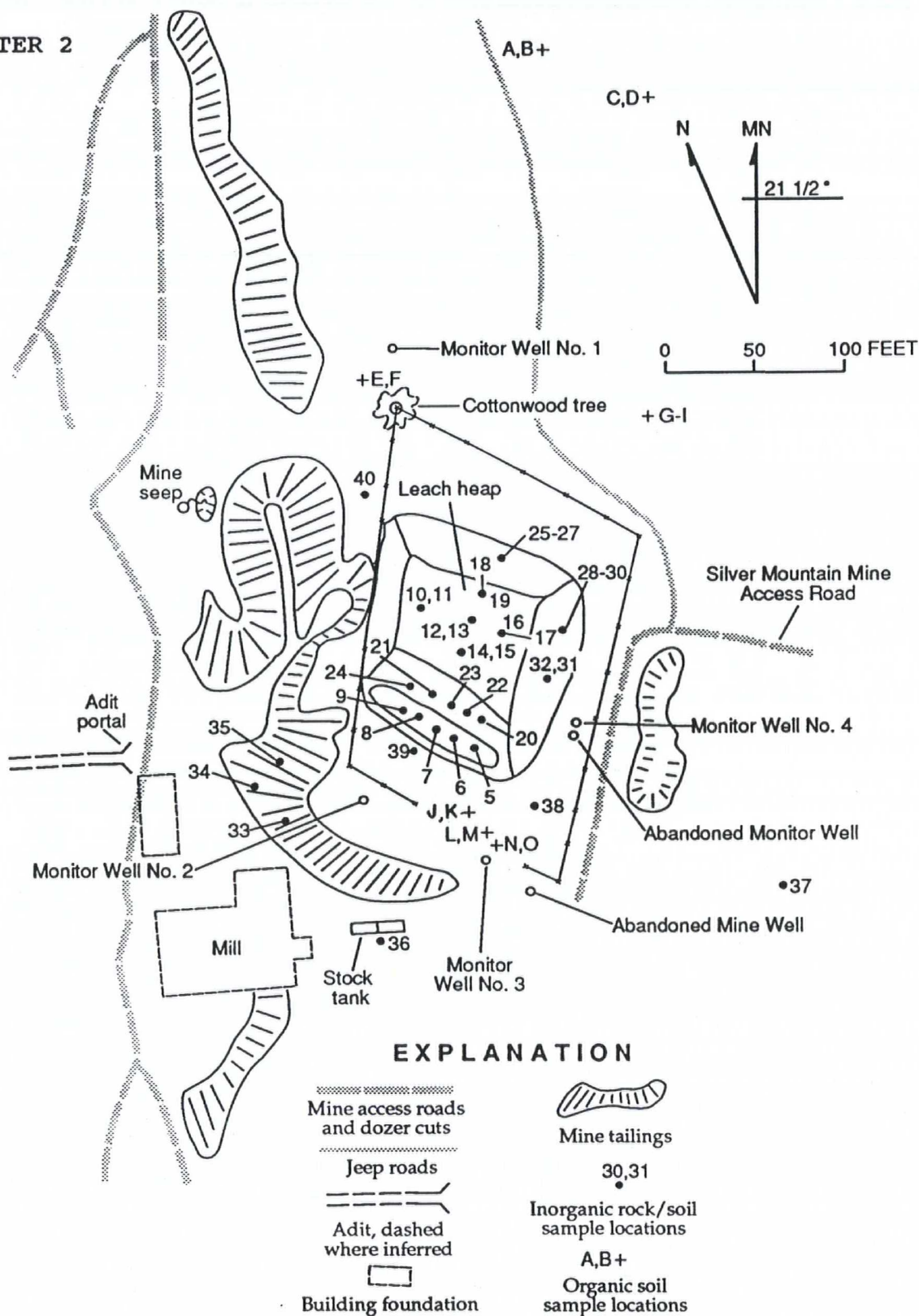


Figure 2.3a. Sample locations for leach heap, mine dump, and nearby soil.

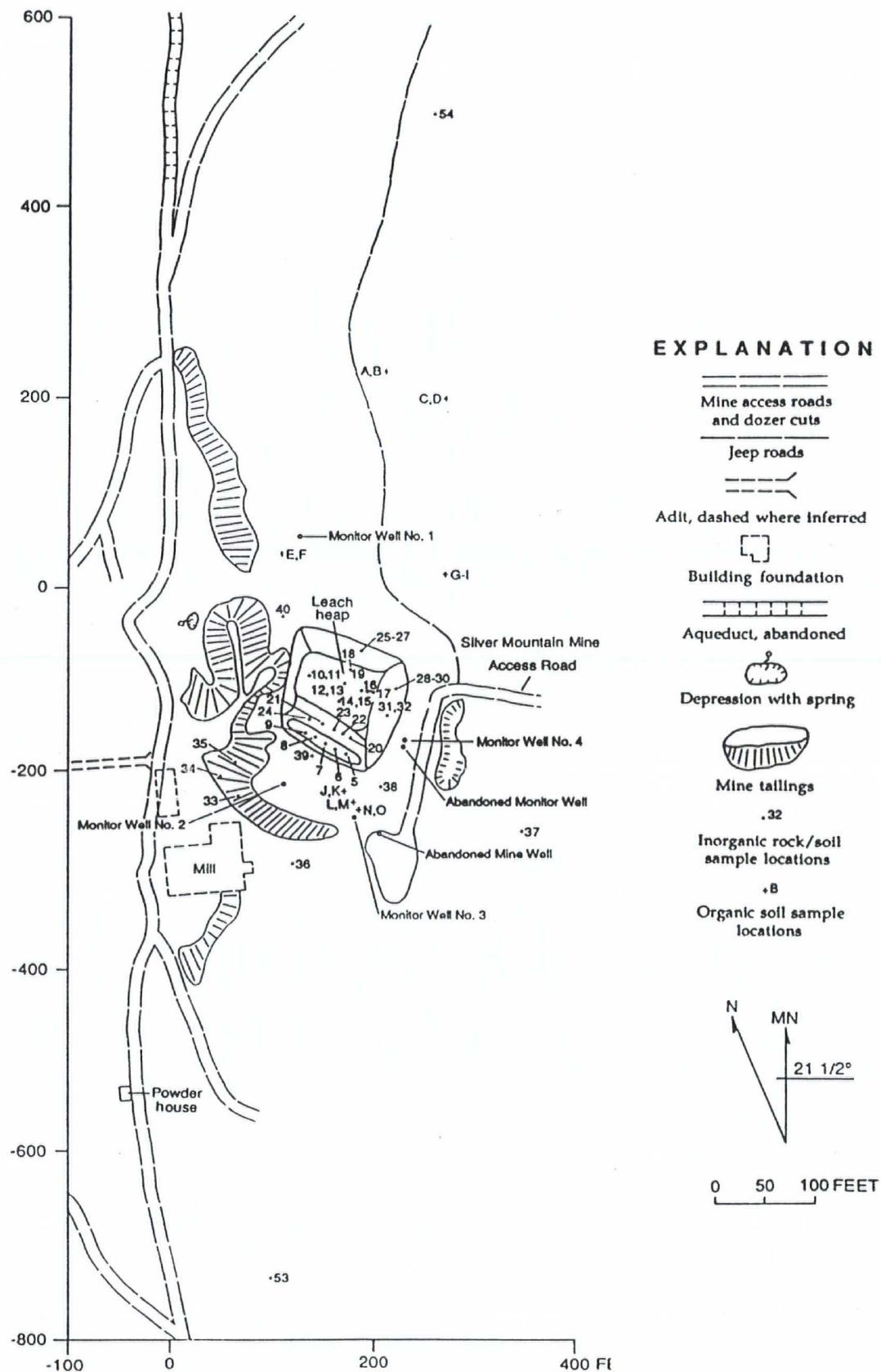


Figure 2.3b. Sample locations for leach heap, mine dump, nearby soil, and background soil.

TABLE 2.5 - LOCATIONS OF LEACH HEAP AND MINE DUMP SAMPLES FOR
CYANIDE AND METALS ANALYSIS

SAMPLE NO.	LOCATION DESCRIPTION
5	Under collection pond liner
6	Under collection pond liner
7	Under collection pond liner
8	Under collection pond liner
9	Under collection pond liner
10	Top of leach heap 5-ft from surface
11	Top of leach heap 2-ft from surface
12	Top of leach heap 5-ft from surface
13	Top of leach heap 2-ft from surface
14	Top of leach heap 5-ft from surface
15	Top of leach heap 2-ft from surface
16	Top of leach heap 5-ft from surface
17	Top of leach heap 2-ft from surface
18	Top of leach heap 5-ft from surface
19	Top of leach heap 2-ft from surface
20	Southern toe of leach heap above pad liner
21	Southern toe of leach heap above pad liner
22	Southern toe of leach heap above pad liner
23	Southern toe of leach heap above pad liner
24S	Southern toe of leach heap above pad liner
25	Base of leach heap north side, 2.5-ft below heap surface
26	DUPLICATE OF 25
27	Base of leach heap north side above pad liner
28	Base of leach heap east side under pad liner
29	DUPLICATE OF 28
30	Base of leach heap eastern side 2.5-ft below surface
31	Base of leach heap eastern side, near liner
32	Base of leach heap eastern side 2.5-ft below surface
33	Mine dump material
34	Mine dump material
35	Mine dump material

Briefly, the 31 samples from the leach heap and mine dump were collected using the following procedures:

Approximately 0.2 ft³ of rock/soil were collected from each location with a stainless steel hand trowel, placed into a large stainless steel pan, and mixed with the trowel.

The material was then screened through nested Tyler sieve screens, sizes 2, 4, 6, and 8, to obtain material less than 2.38 mm in diameter for analysis. A visual estimation of each size fraction was noted.

Materials less than 2.38 mm then went through a stainless steel sample splitter to obtain 250 mL of sample.

Samples were immediately cooled and shipped via overnight delivery to the laboratory for analysis of the parameters listed in Table 2.1. Samples 5, 20, 21, and 28 were also subjected to the EP-Toxicity laboratory test. Samples 8, 12, 22, and 23 were subjected to TCLP. After the laboratory inorganic results were received, the two heap samples with the highest arsenic levels were submitted for petrographic analysis.

2.4.2 Leach Heap and Mine Dump Physical Parameters

To determine the volume of material in the leach heap, points on the heap were included in the site survey. Due to the relative symmetry of the heap, the volume was obtained by considering the heap as a rectangle surrounded by four prisms. The volume of each portion was determined and totalled to derive a heap volume of 111,000 cubic feet. The accuracy of this procedure is estimated at plus or minus 5 percent.

The density of the material on the heap was estimated by filling three 5.5 gallon buckets with heap materials, weighing their contents, and averaging the results. Using this procedure, an average density of 95 lb/ft³ (21 cubic feet per ton) was obtained. The accuracy of this method is estimated at plus or minus 10 percent. Using the volume and density values, the heap contains approximately 5300 tons of material.

The volume of the mine dump, estimated from site survey information, is approximately 108,600 cubic feet. Since the leach heap is made of mine dump materials that were moved onto the leach pad, the densities are assumed to be similar. Thus, approximately 5,200 tons of material make up the mine dump.

Additional investigation of the leach heap and mine dump material included visual estimates of particle size distribution

for each sample, and three semi-quantitative particle size determinations for use in particulate emission determination. Results are presented in Chapters 3 and 4.

Material for the particulate emissions test was obtained from three 75-lb samples of leach heap material. Samples were collected at three different locations, each at a different depth, to allow for variations in the heap with location and exposure to the elements. The particle size distribution test examined materials less than 1 cm in diameter, approximately 20 percent of the total sample volume. A 30 cm X 30 cm X 1 cm volume of this material was used for each analysis. The material was dried for 24 hours at 175 degrees F, placed in a series of nested screens, and processed for 20 "taps" in a Tyler Model B Sieve Screen Shaker, "RO-TAP."

2.5 SOILS INVESTIGATION

Sampling of background soils and soils in areas where spills of organic or inorganic contaminants may have occurred further characterized the nature and extent of contamination. A subset of soil samples was analyzed for the organic contaminants listed on Table 2.3, while others were analyzed for cyanide and metals.

2.5.1 Cyanide and Metals Analysis

Seven soil samples were collected for metals and cyanide analysis. The sample locations are shown in Figure 2.3b, and the locations are briefly described in Table 2.6. Composite sample Nos. 38, 39, 40 were taken in the locations shown on Figure 2.3, where soils surrounding the leach heap may have received runoff or spills of metals and cyanide-containing liquids during or after the operation of the leach heap. Soil sample 40 was taken where process solution overspray was considered possible. Sample No. 39 was taken in an area where solution overflow from the collection trench may have occurred. Sample 38 was taken from an area near a buried pressure vessel. To evaluate areas used by cattle, soil sample 36 was taken in the vicinity of the watering trough and composite soil sample 37 was taken in a grazing area southeast of the heap.

To establish local background conditions, sample Nos. 53 and 54 were collected in areas near enough to the site to have similar physical characteristics as onsite soils yet far enough to have avoided contaminated runoff or spills. No. 53 was 500 ft southwest of the heap, while No. 54 was 500 ft northeast of the heap.

Briefly, the composite soil samples were collected using the following procedures. Details of the procedures are included in the field notes in Appendix C.

Equal volumes of material from the top inch of soil were collected from the corners and center of a ten foot grid using a stainless steel trowel. The sample was then placed into a stainless steel pan and composited. The material was screened through nested Tyler sieve screens, sizes 2, 4, 6, and 8, to obtain material less than 2.38 mm for analysis. A visual estimation of each size fraction was noted.

Material <2.38 mm was split into a 250 mL sample using a stainless steel sample splitter. The sample was immediately cooled and shipped via overnight delivery to the laboratory.

Soil samples were analyzed for the inorganic parameters listed in Table 2.2.

TABLE 2.6 - LOCATIONS OF SOIL SAMPLES FOR METALS AND CYANIDE ANALYSIS

SAMPLE NO.	LOCATION DESCRIPTION
36	Surface soil near watering trough
37	Surface soil near cattle grazing area
38	Surface soil 30-ft SE of heap near buried pressure vessel
39	Surface soil adjacent to southern edge of collection trench
40	Surface soil 15-ft from NW corner of heap
53	Background surface soil 500-ft SW of heap
54	Background surface soil 500-ft NE of heap

2.5.2 Organics Analysis

A total of 15 soil samples were analyzed for the organic parameters listed on Table 2.3. Locations of all organic soil samples are designated by letters in Figure 2.3b and are briefly described in Table 2.7. Both grab and composite samples were taken at the surface and at depth at each location. Grab samples were analyzed for volatile organic compounds (VOC). Composite samples were analyzed for base-neutral acid extractables (BNA) and total organic carbon (TOC).

Samples A and B of surface and deep soil were collected 250 feet north of the heap to determine background levels of organic contaminants in the soils. The areas sampled have minimal evidence of soil disturbance, the same soil and geologic characteristics, and little likelihood of having been affected by site operations.

Based on historic aerial photographs, EPA identified additional locations potentially affected by usage or spillage of industrial fluids, including but not limited to fuel. Samples C and D were taken from a refuse dumping area. Samples E and F were taken near a depression where refuse was disposed. Samples G through I were taken near a crumpled 55-gallon drum. Sample I was a duplicate of H collected for data quality assurance.

Photographs of the cyanide heap leach operation taken in 1981 show a petroleum fuel tank truck parked on the site. The purpose of the truck is not known and it was subsequently removed without confirmation of the tanker's function. Samples J through O were taken near where the tank truck was parked.

TABLE 2.7 - LOCATIONS OF SOIL SAMPLES FOR ORGANICS ANALYSIS

SAMPLE NO.	LOCATION DESCRIPTION
A	Surface soil (Background) 250-ft N of heap
B	Deep soil (Background) 250-ft N of heap
C	Surface soil, refuse area NE of heap
D	Deep soil, refuse area NE of heap
E	Surface soil, NW of heap
F	Deep soil, NW of heap
G	Surface soil, drum N of heap
H	Deep soil, drum N of heap
I	DUPLICATE OF H
J	Surface soil, S of heap
K	Deep soil, S of heap
L	Surface soil, S of heap
M	Deep soil, S of heap
N	Surface soil, S of heap near truck site
O	Deep soil, S of heap near truck site

The organic surface soil sampling procedures are briefly described below.

Surface soil composite - Equal volumes of material from the top inch of soil were collected from the corners and center of a 3 foot grid using stainless steel spoons. The sample was then placed into a stainless steel pan and mixed to achieve homogeneity. Two 8-ounce glass vials, for BNA and TOC analyses, were then filled with composite material.

Surface soil grab - A surface soil grab sample for VOC analysis was collected from the center of the grid and placed in a 4-ounce glass jar using a stainless steel spoon. Deep soil composite - Using a 4-inch diameter stainless steel hand auger, 4-inch increments that extend from about 2" to 6", 10" to 14", 18" to 22", 26" to 30" and 34" to 38" were extracted where possible from a borehole at the center of the surface composite sample grid. The core volumes were placed into a stainless steel pan and composited. Two 8-ounce glass vials, for BNA and TOC analyses, were filled with composited material.

Deep soil grab - From the lowest point of the above borehole, sufficient material was extracted to fill a 4-ounce glass vial for VOC analysis.

Samples were immediately cooled after collection and shipped via overnight delivery to the appropriate contract laboratories for analysis.

2.6 METEOROLOGICAL INVESTIGATION

Meteorological conditions at the site are chiefly of interest in evaluating air transport of contaminants prior to and during remediation. Meteorological conditions at the site were not determined in the field. Rather, climatological data were obtained from the Standard National Oceanographic and Atmospheric Administration (NOAA) tables for the towns of Omak, Oroville, and Tonasket, Washington. Tonasket, 6 miles southeast of the site, is the nearest of the three towns. The NOAA data for Tonasket are considered comparable to site conditions. Monthly temperature and precipitation records were complete for the period between 1983 and 1987. The yearly temperature and precipitation values were averaged for use in this report.

The Ecology and Environment (1985) report states that the average wind speed is 8 miles per hour to the northeast. Department of Ecology field notes from the site stabilization in

June of 1985 note persistent high winds (estimated at 40 mph). Although no direct measurements of wind speed were made, site visits confirm that wind speed is variable.

2.7 SURFACE WATER INVESTIGATION

A natural seep and a stock watering tank are the only perennial surface waters on site (See Figure 1.2, Site Map). The tank is located outside the barbed wire fence presently surrounding the operations area and is filled by a PVC pipe with water siphoned from saturated lower workings of the abandoned mine. The seep lies in a depression 110 feet northwest of the tailings pile. Since flow from the seep is minimal, no flow rate measurements were taken during the investigation.

To evaluate site surface water conditions, field parameters were measured for the seep and the stock tank, and samples from both were collected for metals, cyanide, and anions analysis. Field measurements and water sampling protocol are discussed in Section 2.9.

Surface water does not travel from onsite to offsite streams, lakes, or other open water. Offsite water was investigated by means of the U.S. Geological Survey (USGS) topographic map (see Figure 2.4) and observations made during site visits between October 1988 and July 1989. According to the USGS map, offsite surface waters include some small marshes and seasonal ponding in low-lying portions of Horse Springs Coulee. The map also indicates a seasonal pond on the ridge west of the site and several small lakes to the north and south. At their nearest point, the ponds and marshes are about one half mile east of the site.

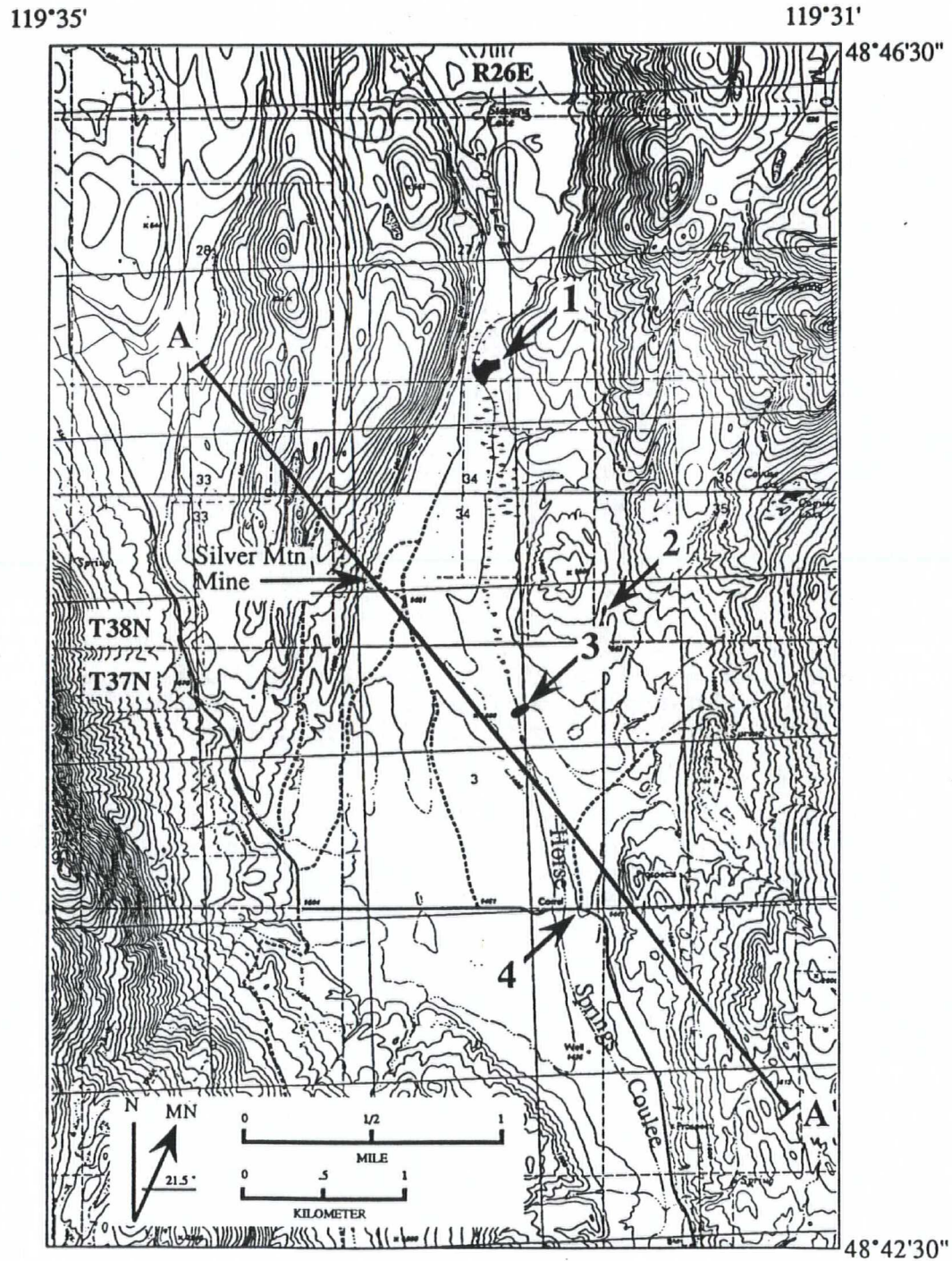


Figure 2.4. Surface water bodies and location of cross section in figure 3.1.

Field observations in spring of 1989 indicated dry basins at the two potential water-impounding features southeast of the site (indicated as 2 and 3 on Figure 2.4). At the same time, the pond to the northeast was about one tenth the size indicated on the USGS map. Water was consistently absent during all site visits at point 4, where the intermittent stream is shown to cross the road (Figure 2.4). Due to absence of surface water near the site and the distance of perennial streams or lakes from the site, offsite surface waters were not sampled.

2.8 GROUND WATER INVESTIGATION

Four monitoring wells were installed around the base of the heap to determine whether contamination from the cyanide leaching operations has reached the ground water. Water from three nearby offsite wells was also sampled. Additional information about the region's ground water was obtained through literature searches.

2.8.1 Offsite Wells

The nearest water supply wells are three large diameter irrigation wells located 2.25, 3.5, and 4 miles south-southeast of the site. The nearest residential well is adjacent to the third irrigation well. Offsite well locations are shown in Figure 2.5.

Based on Russell and Eddy (1972), these wells are considered to be downgradient of the site. The irrigation wells supply water for field sprinkler systems and are driven by above-ground electric pumps. A search of well logs filed with the Washington Department of Ecology located a log for only the furthest irrigation well. This well is 10 inches in diameter and penetrates unconsolidated glacial-fluvial sediments to a total depth of 80 feet. The well yields 150 gallons per minute and has a static water level of 20 feet from the surface.

All four offsite wells draw from the Aeneas Lake-Horse Springs Coulee aquifer (Russell and Eddy, 1972). To determine whether contaminants have migrated into the ground water in the main aquifer, samples were taken from the nearest two irrigation wells and the residential well. The irrigation well adjacent to the residential well is expected to have comparable characteristics. Field measurements and sampling procedures are discussed in Section 2.9.

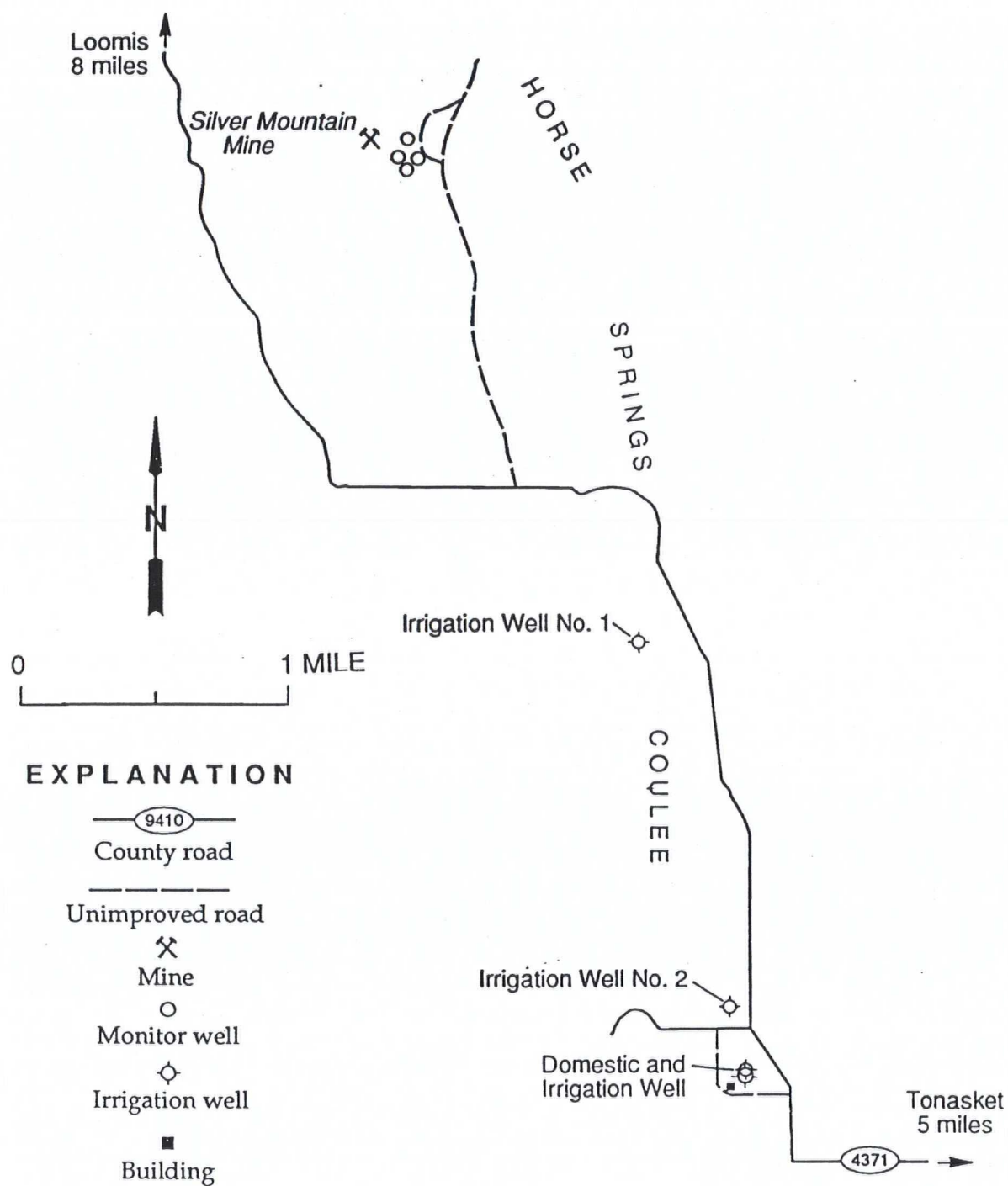


Figure 2.5. Locations of offsite water supply wells.

2.8.2 Onsite Monitoring Wells

Four monitoring wells were installed to characterize the site aquifer. Three of the four monitoring wells were installed in October 1988 and the fourth well was installed in April 1989. The first attempt at a fourth well (Well 4a) was ceased due to difficulties in removing the well casing and the borehole was abandoned and sealed. The completed Well 4 is installed adjacent to the abandoned Well 4a. Monitoring well locations are shown in Figure 1.2 and well diagrams are shown in Figures 2.6 through 2.9.

Three rounds of samples were collected and analyzed for the parameters listed in Table 2.2. Field parameters -- temperature, static water levels, pH, Eh, and electrical conductivity -- were also measured regularly for the monitoring wells. Field parameters and sampling methods are discussed in Section 2.9.

Each monitoring well was purged before sampling using a submersible downhole electric pump, with the exception of Well No. 1. Monitoring Well No.1 was purged with a teflon bailer because very little water was present. Samples were collected from each monitoring well after the static water level was re-established.

2.8.2.1 Monitoring Well Installation

Installation of the monitoring wells was conducted in accordance with guidance under U.S. Environmental Protection Agency (1986). The procedures included:

- Steam cleaning of drill rig, rods, and bits prior to drilling each well
- Use of contaminant-free drill rod thread lubricant
- Use of PVC monitoring well casings and screens with water tight joints
- Use of stainless steel casing centralizers to center well screens in the drillhole.
- Steam cleaning of all well materials (casing, screen, and centralizer)
- Use of commercial washed and sized sand for fill between the casing and the saturated zone, bentonite tablets for fill between the casing and the unsaturated zone, and concrete fill to plug and seal well to the maximum frost line (four feet below the surface)

2.8.2 Onsite Monitoring Wells

Four monitoring wells were installed to characterize the site aquifer. Three of the four monitoring wells were installed in October 1988 and the fourth well was installed in April 1989. The first attempt at a fourth well (Well 4a) was ceased due to difficulties in removing the well casing and the borehole was abandoned and sealed. The completed Well 4 is installed adjacent to the abandoned Well 4a. Monitoring well locations are shown in Figure 1.2 and well diagrams are shown in Figures 2.6 through 2.9.

Three rounds of samples were collected and analyzed for the parameters listed in Table 2.2. Field parameters -- temperature, static water levels, pH, Eh, and electrical conductivity -- were also measured regularly for the monitoring wells. Field parameters and sampling methods are discussed in Section 2.9.

Each monitoring well was purged before sampling using a submersible downhole electric pump, with the exception of Well No. 1. Monitoring Well No.1 was purged with a teflon bailer because very little water was present. Samples were collected from each monitoring well after the static water level was re-established.

2.8.2.1 Monitoring Well Installation

Installation of the monitoring wells was conducted in accordance with guidance under U.S. Environmental Protection Agency (1986). The procedures included:

- Steam cleaning of drill rig, rods, and bits prior to drilling each well
- Use of contaminant-free drill rod thread lubricant
- Use of PVC monitoring well casings and screens with water tight joints
- Use of stainless steel casing centralizers to center well screens in the drillhole.
- Steam cleaning of all well materials (casing, screen, and centralizer)
- Use of commercial washed and sized sand for fill between the casing and the saturated zone, bentonite tablets for fill between the casing and the unsaturated zone, and concrete fill to plug and seal well to the maximum frost line (four feet below the surface)

- Construction of an outwardly sloping concrete well pad
- Use of locking metal well cap stamped with location and pertinent information and use of a key padlock.

To ensure a stable and sediment-free ground water flow, a submersible 220 volt pump was temporarily installed into each well to purge several well volumes. Well 1 was purged with a teflon bailer, as it contained very little water.

2.8.2.2 Slug Test

A slug test was conducted on Well 3 in April of 1989 to measure the hydraulic conductivity of the aquifer. This method was selected as best suited to the site's hydrogeology due to the low water yield observed during well construction and the aquifer's medium of glacial drift (a mixture of gravel, sand, silt, and clay).

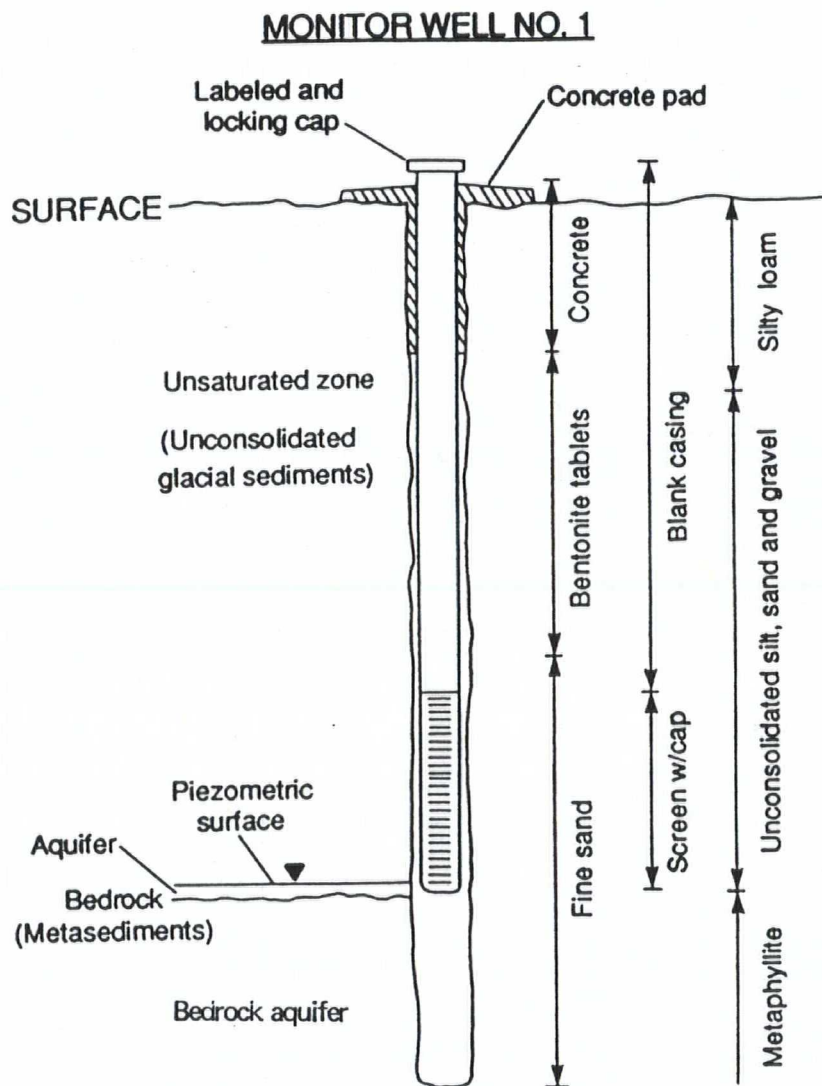
The slug test involved removing the water from the well and measuring the water level recovery at closely spaced time intervals. Well 3 recovered to its initial water level in approximately 56 hours. The data were plotted on semilog paper, and the lag time was obtained using the initial rate of inflow following methods of M.J. Hvorslev (1951). Data from the slug test are included in Appendix C, and the resulting hydraulic conductivity and ground water flow values are discussed in Section 3.6.1.

2.9 WATER SAMPLING PROCEDURES

This section describes the procedures used to measure physical and chemical parameters in the field and to collect samples for laboratory analysis. These procedures were followed for all water samples, including onsite monitoring wells, offsite wells, and onsite surface water, unless otherwise stated. Results are discussed in Chapter 3.

2.9.1 Static Water Levels

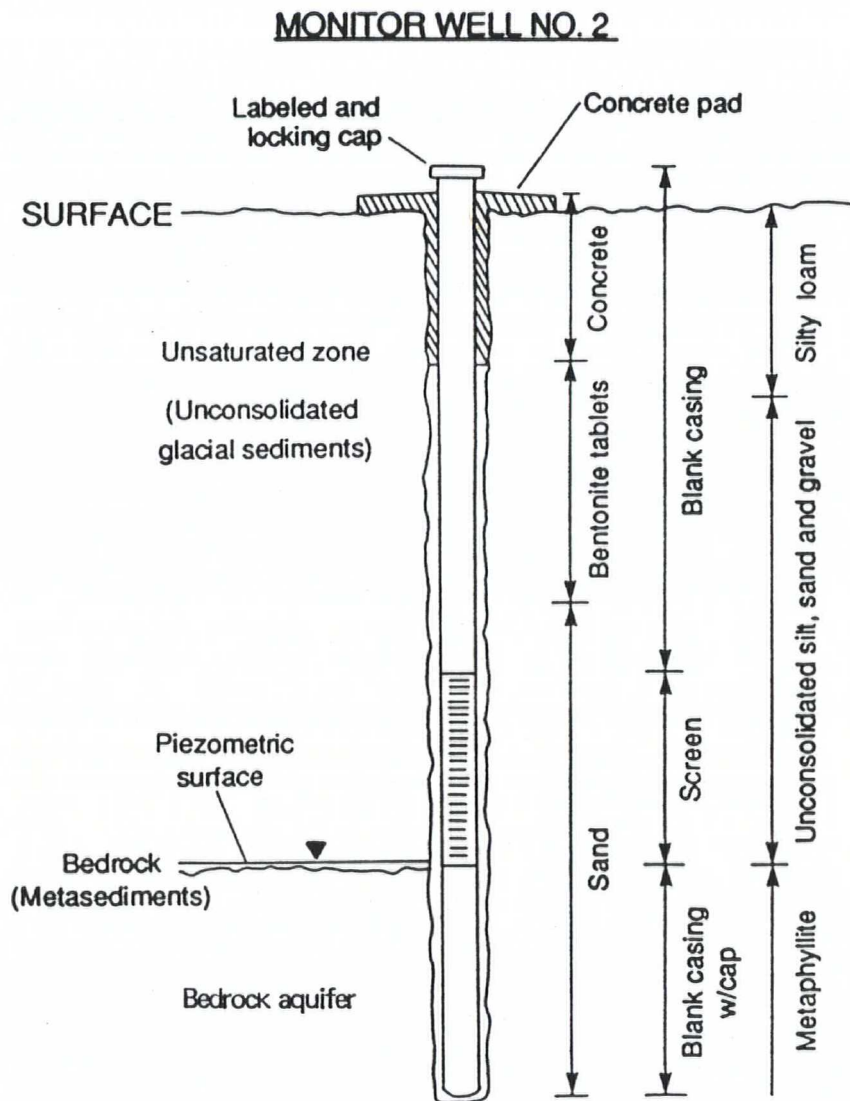
Static water levels (SWL) were measured prior to sampling each monitoring well. An electrical depth probe calibrated in feet was used to measure the static water level relative to a measuring point on the well casing. Static water levels were then converted to elevations in feet above mean sea level based on surveyed well-head elevations. The water elevations were used to generate ground water surface diagrams.



Location: Silver Mountain Mine, Tonasket, WA
Well No: 38/26-34 N 1

0 5 10 FEET

Figure 2.6. Construction and lithologic diagram for well 1.



Location: Silver Mountain Mine, Tonasket, WA
Well No: 38/26-34 N2

0 5 10 FEET

Figure 2.7 Construction and lithologic diagram for well 2.

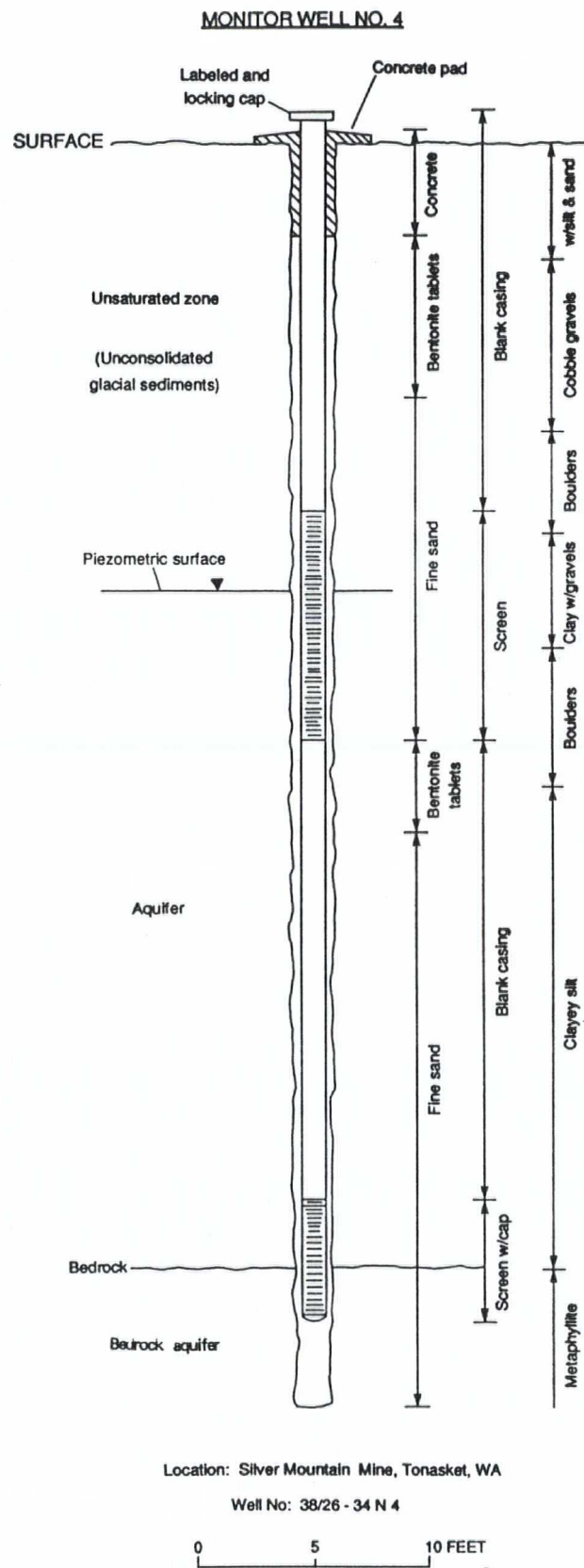


Figure 2.8 Construction and lithologic diagram for well 3.

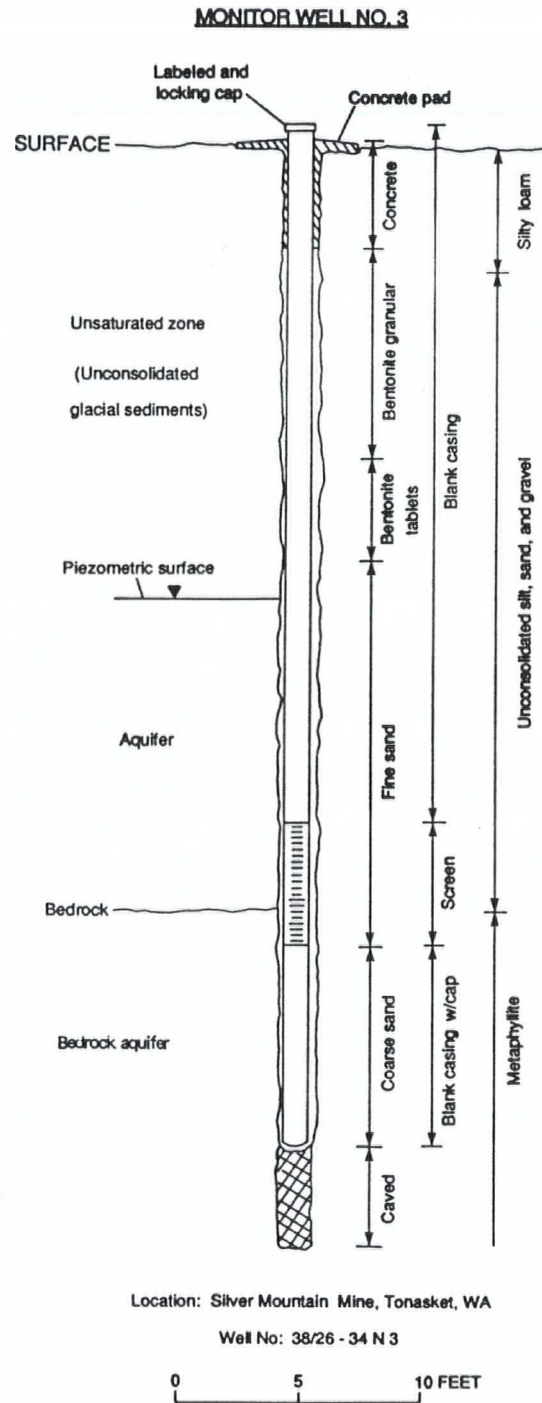


Figure 2.9 Construction and lithologic diagram for well 4.

2.9.2 Temperature

Water temperatures were measured with a temperature probe connected to an Orion model SA-250 meter. To measure onsite ground water temperatures, the probe was lowered into the monitoring wells to a depth where the temperature was uniform and stable. The temperature and depth probes and associated lines were thoroughly rinsed between each reading with distilled deionized water. Temperatures of offsite wells and onsite surface water were measured by insertion of the probe in a teflon sampling bottle immediately after sample collection.

2.9.3 Eh, pH, Electrical Conductivity

Eh and pH were measured in the field with a platinum Eh electrode and a Ross pH electrode, respectively, and the Orion model SA-250 meter. Electrical conductivity was measured with an Orion model SL-1 conductivity meter with a built-in temperature sensor. The conductivity meter automatically compensated for temperature to 25° Celsius, with an accuracy of 2% per °C.

2.9.4 Sampling Collection and Preservation

Sampling procedures for the four onsite wells, three offsite wells, and two surface water locations were as follows. A teflon bailer was rinsed with deionized water three times. To measure the effectiveness of the bailer rinse before sampling, additional deionized water was flushed through the rinsed bailer, collected, and preserved as a rinsate sample for quality control.

Water from the sample source was then collected with the bailer and used to rinse five pre-cleaned, one-liter Nalgene sample bottles. Finally, water was poured directly into each bottle from the 250 mL bailer. A number of bailer volumes were required to collect an adequate amount of sample from the monitoring wells.

Samples from offsite water supply wells were collected from valves near the well heads after allowing enough water to discharge through the valve to reach steady temperature. Samples from the onsite surface water, the stock tank and seep, were collected as grab samples.

One of the five liters for each sample was used for field tests and field parameters. The water was checked for pH, Eh, and electrical conductivity, as described in section 2.9.3. Colorimetric spot tests were then conducted to determine whether sulfides or oxidizing agents were present in sufficient quantities to interfere with cyanide analysis. Spot tests were conducted in accordance with the aqueous cyanide preservation

protocol adopted by EPA Region 10 Regional Quality Assurance Management Office (Jones, 1986). Free cyanide was also determined in the field, using the EM Merck 10027-1 cyanide test kit. This method utilizes cyanide sensitive colorimetric test paper and is accurate to 1 ppm.

Tables 2.2 and 2.8 summarize the analytical parameters and field procedures, respectively, for the remaining four liters collected at each location. All samples were filtered and preserved as needed within a few minutes of sampling, except for Round 1 metals which were preserved with nitric acid after receipt by the laboratory.

The first liter of each sample collected for laboratory analysis was for dissolved metals and was filtered to remove suspended material. All filters were 47 mm in diameter and included a prefilter, a 0.8 μ m membrane, and a 0.45 μ m final filter. The filters were prepared by pumping two or three system volumes of distilled water followed by two or three system volumes of sample water through the entire filter system. The water pump used was a variable speed 12-volt pump with nylon gears and teflon and polypropylene components. Interconnecting hoses between the pump and the filter assembly were made of teflon. The sample was finally preserved by addition of nitric acid to a pH <2.

The second liter of sample was not filtered and was used to determine total metals. This sample was also preserved with nitric acid. The third liter was collected for cyanide (total and weak acid dissociable) analysis and was preserved with sodium hydroxide to a pH of 12. The fourth liter, for anion analysis, was not filtered. All samples were immediately cooled and sent to the laboratory for analysis.

TABLE 2.8 - SUMMARY OF FIELD PROCEDURES FOR WATER SAMPLES

1st liter: DISSOLVED METALS	filtered, preserved with nitric acid and cooling
2nd liter: METALS	unfiltered, preserved with nitric acid and cooling
3rd liter: CYANIDE (Total and WAD)	unfiltered, preserved with sodium hydroxide
4th liter: ANIONS	unfiltered, preserved by cooling

2.10 DEMOGRAPHIC INVESTIGATION

A review of demographic information pertaining to Silver Mountain Mine is found in section 6.3.1.

2.11 ECOLOGICAL INVESTIGATION

The approach to the ecological investigation is presented in section 7.3.1.

REFERENCES

- American Public Health Association, 1985, Standard methods for the examination of water and wastewater, 16th edition: Washington, DC, 1134 p.
- Hvorslev, M.J., 1951, Time lag and soil permeability in ground-water observations: U.S. Army Corps of Engineers, Waterways Experiment Station, Bulletin 36, 49 p.
- Jones, R.R., 1986, Aqueous sample cyanide preservation; acceptance and stipulation: U. S. Environmental Protection Agency Region 10 memorandum, November 4, 1986, Seattle.
- Russell, Robert H. and Eddy, Paul A., 1972, Geohydrologic evaluation of Aeneas Lake-Horse Springs Coulee, Okanogan County, Washington: Washington Department of Ecology Investigations, January 1972, 16 p., 1 app.
- U.S Environmental Protection Agency, 1986, RCRA ground-water monitoring technical enforcement guidance document: Publication no. OSWER-9950.1, 208 p., 3 app.

CHAPTER 3 - PHYSICAL CHARACTERISTICS OF THE STUDY AREA

This chapter discusses the results of the fieldwork conducted to determine the physical characteristics of the study area. Results of laboratory analyses are discussed in Chapter 4.

3.1 TOPOGRAPHY

The Silver Mountain Mine site lies on the western edge of Horse Springs Coulee. The Coulee was originally formed by southward flowing meltwater from Pleistocene continental glaciers. It is characterized by a wide, glacial outwash plain. The Coulee's western border is a north-trending ridge of exposed bedrock. The mine workings were developed in this ridge. Mined materials were piled near the mine entrance in a narrow valley separated by a low limestone ridge from the main portion of the Coulee. Elevations near the mine range from 1480 feet above mean sea level in the Coulee to 2120 feet at the high point of the ridge. The topography of the area is indicated in Figure 2.1. Geology is shown in Figure 2.2.

3.2 CLIMATE

The climate of the area is generally characterized by very cold winters and warm summers, due to its northern latitude east of the Cascade Mountains and just south of the Canadian border. The climatological data stations nearest to the mine site are in the towns of Omak, Oroville, and Tonasket, Washington. These data are summarized below. Based on available monthly precipitation records (Bureau of Reclamation written communication, 1989), annual precipitation ranges from 9.61 to 19.60 inches. Data for the town of Oroville, 18 miles to the north show a mean annual precipitation of about 11.4 inches. Most precipitation occurs as rain from March to May and in September. Snowfall occurs in November and December.

The yearly average temperature is approximately 50°, with highs in the mid-seventies in July and August and lows in the lower twenties during December and January. During an average winter the top three feet of soil are generally frozen, but in severe winters the frost level can drop to as much as four feet below the surface (L. Andrews, University of Washington Cooperative Extension, Okanogan County, personal communication, 1988).

Ecology and Environment (1985) reported a mean annual wind speed of 8 miles per hour to the northeast. Maximum winds in

excess of 30 mph were observed during site stabilization in 1982 and during the Remedial Investigation in 1989.

3.3 GEOLOGY

3.3.1 Regional Geology

The following discussion of regional geology is based on geologic reports by Rinehart and Fox, (1972, 1976).

Silver Mountain Mine lies on the eastern flank of the northern Cascade Mountain orogenic province. The area is characterized by numerous Jurassic and Cretaceous age plutons composed of granite, quartz diorite, quartz monzonite, and granodiorite. The emplacement of these plutons has metamorphosed the overlying strata into a metasediment complex of Permian to Triassic age.

Folding and faulting are generally seen in north to northwest trends. Many of these folds and faults resulted from intense periods of deformation during intermittent plutonic emplacement. Large scale metamorphism occurred from the Permian through the Triassic, while additional folding and faulting occurred during the Cretaceous.

3.3.2 Local Geology

This interpretation of the local geology is based on the field work described in Section 2.2. A map of site geology is shown in Figure 2.2.

The mine workings are within the Permian Anarchist Group (Lexicon of Geologic Names of the United States, USGS Bulletin 1200, p 92.). Rocks found here include metaschist, metaphyllite, and metaquartzite. Interwoven and outcropping within this group is the Triassic/Permian Mafic Intrusive Rock which occurs locally as massive amphibolite, greenstone, and metadiabase. These intrusive outcrops are outside the area mapped by the Bureau. The parent rocks of the amphibolite, greenstone, and metadiabase were intensely metamorphosed, probably by the Triassic Loomis Pluton found one mile west of the mine.

A small outcrop of metaquartzite occurs near the southernmost limestone outcrop in the mapped area. It is very hard and dense and is characterized by conchoidal fracturing, a type of fracturing that gives a smoothly curved surface such as that seen in obsidian or quartz. Some small pyrite crystals were observed, which could account for the local iron staining.

About 100 yards east of the mine entrance lies an isolated outcrop of hard, dense limestone. It strikes to the northeast as a low-lying ridge following the regional strike in the other metasediments. The age of the limestone and its geologic relationship with the other rocks are unknown.

Between the Anarchist Group/Mafic Intrusive Rock and the limestone is a veneer of Pleistocene Glacial Drift. This is the type of material directly underlying the leach heap pad (Figure 3.1). The glacial drift is composed of silt, sand, and gravel, and it is found throughout the Horse Springs Coulee area. The thickness of the glacial drift varies, but monitoring well logs showed a range of 14 to 49 feet in the leach heap area. To the east and south, in the main channel of the coulee, the thickness of the glacial drift layer is much greater due to its location in the center of the glacial basin. Russell and Eddy (1972) report the thickness of glacial drift to range from 20 to 150 feet with an average of about 100-125 feet in Horse Springs Coulee.

Folds in the Anarchist Group strike northerly, with a 71 to 82 degree easterly dip. The internal foliation of the Anarchist Group also trends north 10 degrees east and inclines 84 degrees to the east. The limestone strikes to the north with a 55 to 86 degree easterly dip. These stratigraphic and structural relations further substantiate that the source of metamorphism and deformation was the Loomis Pluton.

Two faults were mapped at the site. They occur within the limestone ridge and appear to be right-lateral slip faults. In addition, a large thrust fault trending north with an eastward moving upward plate has been inferred by Rinehart and Fox (1972, 1976) and tentatively mapped adjacent to the mine workings. Since its actual occurrence and position is speculative, it is not shown on the geologic map (Figure 2.2).

3.4 SOILS AND LEACH HEAP MATERIALS

As discussed in Section 2.4 and 2.5, soils, mine dump and leach heap materials were sampled for laboratory analysis. Table 3.1 summarizes field data gathered during the collection of samples for metals and cyanide analysis. Samples were screened with Tyler sieve screens of 9.42, 4.76, 3.36, and 2.38 mm mesh openings. Materials less than 2.38 mm in size were used in the chemical analyses. The percentage by volume of materials remaining on each sieve screen was estimated visually, and sample lithology was described. The sample lithology includes rock/soil type, color, amount of hydrothermal alteration, and field identification of sulfide mineralization. This information is included in Table 3.1.

TABLE 3.1 - INORGANIC ROCK/SOIL SAMPLING FIELD DATA

SAMPLE NUMBER	SIZE DISTRIBUTION (mm), WITH PERCENT PASSING					SAMPLE DESCRIPTION
	>9.42	≤9.42>4.76	≤4.76>3.36	≤3.36>2.38	≤2.38	
5	20	25	15	15	25	Under collection pond liner, unaltered, green schist-phylite, no mineralization
6	10	20	10	10	50	Under collection pond liner, unaltered, green schist-phylite, no mineralization
7	3	5	1	1	90	Under collection pond liner, fined grained (clayey) brown soil, no mineralization
8	10	20	10	10	50	Under collection pond liner, unaltered green schist-phylite, no mineralization
9	10	20	10	10	50	Under collection pond liner, unaltered, green schist-phylite, no mineralization
10	50	10	7.5	7.5	25	Top of heap, 5-ft from surface, altered schist-phylite, <5% quartz, no visible sulfides
11	50	10	7.5	7.5	25	Top of heap, 2-ft from surface, 70% unaltered phyllite, 30% highly altered schist-phylite, <5% quartz, no visible sulfides
12	50	10	5	5	35	Top of heap, 5-ft from surface, altered schist-phylite, <5% quartz, no visible sulfides
13	50	10	5	5	35	Top of heap, 2-ft from surface, 70% unaltered phyllite, 30% highly altered schist-phylite, no visible mineralization
14	20	30	15	15	20	Top of heap, 5-ft from surface, 70% unaltered schist-phylite, 30% altered schist-phylite with quartz containing disseminated sulfides, possibly chalcopryrite and arsenopryrite.
15	30	20	15	15	20	Top of heap, 2-ft from surface, 70% unaltered schist-phylite, 30% altered schist-phylite with quartz containing disseminated sulfides, possibly chalcopryrite and arsenopryrite.
16	30	30	<10	>10	20	Top of heap, 5-ft from surface, 70% unaltered schist-phylite, 30% altered schist-phylite with quartz containing disseminated sulfides, possibly chalcopryrite and arsenopryrite.
17	30	30	10	10	20	Top of heap, 2-ft from surface, 70% unaltered schist-phylite, 30% altered schist-phylite with quartz containing disseminated sulfides, possibly chalcopryrite and arsenopryrite.
18	20	30	10	10	30	Top of heap, 5-ft from surface, 70% unaltered schist-phylite, 30% altered schist-phylite with quartz containing disseminated sulfides, possibly chalcopryrite and arsenopryrite.

TABLE 3.1 (cont'd). - INORGANIC ROCK/SOIL SAMPLING FIELD DATA

SAMPLE NUMBER	SIZE DISTRIBUTION (mm), WITH PERCENT PASSING					SAMPLE DESCRIPTION
	>9.42	≤9.42>4.76	≤4.76>3.36	≤3.36>2.38	≤2.38	
19	30	30	10	10	20	Top of heap, 2-ft from surface, 70% unaltered schist-phylite, 30% altered schist-phylite with quartz containing disseminated sulfides, possibly chalcopryrite and arsenopryrite.
20	60	15	5	5	15	Southern toe of heap, above pad liner, <5% unaltered schist-phylite, 85% very altered schist-phylite, 10% quartz with disseminated chalcopryrite and other sulfides.
21	40	20	10	10	20	Southern toe of heap, above pad liner, <5% unaltered schist-phylite, 85% very altered schist-phylite, 10% quartz with disseminated chalcopryrite and other sulfides.
22	30	20	10	10	30	Southern toe of heap, above pad liner, 70% altered to very altered schist-phylite, 30% unaltered schist-phylite, <5% iron-stained quartz with unidentifiable disseminated sulfides.
23	30	20	10	10	30	Southern toe of heap, above pad liner, 70% altered to very altered schist-phylite, 30% unaltered schist-phylite, <5% iron-stained quartz with unidentifiable disseminated sulfides.
24	30	25	10	10	25	Southern toe of heap, above pad liner, 80% unaltered schist-phylite, 20% altered phyllite, no visible mineralization.
25	20	30	10	10	30	Base of heap, northern side, 2.5-ft below heap surface, unaltered phyllite, no mineralization.
26	20	30	10	10	30	DUPLICATE OF 25
27	20	30	10	10	30	Base of heap, northern side, above pad liner, unaltered phyllite, no mineralization.
28	20	30	10	10	30	Base of heap, eastern side, under pad liner, weathered, iron-stained phyllite with 10% altered phyllite.
29	20	30	10	10	30	DUPLICATE OF 28
30	20	30	15	15	20	Base of heap, eastern side, 2.5-ft below heap surface, 95% altered to very altered schist-phylite, <5% quartz with unidentifiable disseminated sulfides.
31	40	15	15	5	25	Base of heap, eastern side, proximity of liner, altered and very altered phyllite mixed with weathered, iron-stained schist-phylite, no mineralization.
32	20	30	15	15	20	Base of heap, eastern side, 2.5-ft from heap surface, altered to very altered schist-phylite, <2% quartz with very disseminated unidentifiable sulfides.

TABLE 3.1 (cont'd). - INORGANIC ROCK/SOIL SAMPLING FIELD DATA

SAMPLE NUMBER	SIZE DISTRIBUTION (mm), WITH PERCENT PASSING					SAMPLE DESCRIPTION
	>9.42	≤9.42>4.76	≤4.76>3.36	≤3.36>2.38	≤2.38	
33	20	30	15	15	30	Mine dump material, very altered, iron-stained schist-phyllite with <10% quartz, no visible mineralization.
34	30	25	10	10	25	Mine dump material, very altered, iron-stained schist-phyllite with 40% very altered, iron-stained quartz containing unidentifiable sulfides.
35	10	30	10	10	40	Mine dump material, unaltered green phyllite with <1% white quartz, no mineralization.
36	5	5	5	5	80	Surface soil, near cattle watering trough, primarily brown soil with high organic composition.
37	0	0	0	0	100	Surface soil, black loamy soil.
38	30	20	10	10	30	Surface soil, 30-ft SE of heap, proximate to buried pressure vessel, altered, iron-stained phyllite.
39	10	20	10	10	50	Surface soil, adjacent to southern edge of collection pond, altered, iron-stained phyllite.
40	10	10	5	5	70	Surface soil, 15-ft from NW corner of heap, unaltered, green-gray phyllite.
53	<5	5	<5	<5	85	Surface soil, 500-ft SW of heap, colluvium from ridge to the west, weathered brown-green phyllite.
54	<2	5	<5	5	85	Surface soil, 500-ft NE of heap, colluvium from ridge to the west, weathered tan-brown phyllite.

As would be expected, the field data generally indicate an overall predominance of gravel-sized pieces of metamorphic bedrock such as greenschist, phyllite, limestone in samples from the heap and mine dump, reflecting their origin in the mine. The surface soil is finer grained glacial drift and colluvium, composed of eroded bedrock material that originated in the area extending from the mine site northward into Canada.

Table 3.2 gives the results of particle size distribution tests conducted on three samples from the pile. This information is primarily for use in estimating current and future particulate emissions, and emissions during remedial activities.

Table 3.2. Particle size distribution for determination of particulate emissions. All weights are in grams, all sizes are Tyler Sieve Screen sizes.

Sample Depth	Total Weight	Particle Size and Estimated Percent						
		>5	<5	<9	<14	<32	<65	<200
1-5 ft	1230.5 100%	529.0 43%	219.7 18%	132.5 11%	174.4 14%	92.9 7%	56.9 5%	25.1 2%
5-10 ft	1775.3 100%	1005.8 57%	286.9 16%	131.9 7%	155.1 9%	93.2 5%	72.7 4%	30.1 2%
10-15 ft	997.6 100%	338.8 34%	162.6 16%	121.2 12%	179.2 18%	105.1 11%	70.5 7%	20.2 2%

3.5 HYDROLOGY AND HYDROGEOLOGY

This section characterizes the physical conditions of ground and surface water at and near the site based on field measurements described in Chapter 2. It includes a description of the aquifers underlying the site and a discussion of field results.

3.5.1 Surface Water Hydrology

In Section 2.7, Surface Water Investigation, it was noted that the only perennial surface waters at the Silver Mountain Mine site are a natural seep northwest of the leach heap and a stock tank filled by water piped from the abandoned mine. These are indicated on the site map, Figure 1.2. Offsite, the only perennial surface water in a three mile radius of the site is a Stevens Lake, a shallow lake 1.75 miles northeast of the site. The lake is used for stock watering and fishing.

The nearest river is the Okanogan, about 5 miles east of the site. Other bodies of water near the site include Spectacle Lake (4 miles north of the site), Whitestone Lake (4.5 miles northeast), Lemanasky Lake (3.5 miles southeast), and Aeneas Lake (4.5 miles southeast). These lakes are not connected to the site by any surface water body and are not shown on site maps.

The natural seep lies in a shallow depression about 100 feet northwest of the leach heap. Its flow is minimal and was not measured during the RI. The seep fills a shallow pool about 20 square feet in area and 6 inches deep in the center. The size of the pool diminishes in the summer. No surface runoff from the seep was observed.

The 400 gallon stock watering tank is located outside the fenced area. It is filled with water siphoned from a saturated portion of the mine workings. The PVC pipe emerges from the mine portal approximately 20 feet above the base of the leach heap. Estimated flow is less than one gallon per minute.

The heap is located on a slight rise in the area. Runoff tends to flow away from the heap to the north and south. Some runoff from the covered heap collects in the leachate pond in the south edge of the heap where the top liner extends into the pond basin.

3.5.2 Site Aquifer

Ground water conditions at the Silver Mountain Mine site may be generally described in terms of two aquifers, a shallow aquifer in the unconsolidated glacial drift and a bedrock aquifer in fractured zones in the bedrock below the drift. These relationships are shown in Figure 3.1 by cross sections. The locations of the cross sections are shown in Figure 2.1 and 2.4. The regional cross section, A-A', depicts the relationship of the site to the adjacent hills and to Horse Springs Coulee. The site proper is shown in cross section B-B'.

The shallow aquifer underlying the leach heap is the primary ground water concern. For this reason, the Remedial Investigation concentrates on characterization of ground water in the shallow aquifer. The bedrock aquifer was not investigated in detail.

The same stratigraphic sequence of glacial drift on top of bedrock occurs in Horse Springs Coulee. Russell and Eddy (1972) refer to the aquifer in unconsolidated glacial drift in Horse Springs Coulee as the Horse Springs Coulee-Aeneas Lake aquifer. This aquifer is referred to simply as Horse Springs Coulee aquifer in this report. Water enters the Horse Springs Coulee aquifer from adjacent higher ground and flows south towards Aeneas Lake.

The shallow aquifer at the mine site is on the margin of the Horse Springs Coulee aquifer and is separated from it by a low limestone ridge (Figure 2.1). It is not clear whether the ridge is continuous with the bedrock formation below or is "floating" in the glacial drift layer. The cross sections in Figure 3.1 are drawn assuming that the limestone ridge is part of bedrock. In this report, the term "shallow aquifer" is used to denote the aquifer in glacial drift at the site, as distinct from the central portion of the Horse Springs Coulee aquifer to the east of the limestone ridge.

3.5.2.1 Aquifer Thickness

The shallow aquifer at the site is classified as an unconfined, or "water table" type. Well logs from the four monitoring wells show the thickness of the shallow aquifer underlying the mine site (Figures 2.6 - 2.9). All four monitoring wells completely penetrate the shallow aquifer. At Wells 1 and 2, the shallow aquifer is only a few inches thick, where it pinches out against the bedrock-ridge to the west. At Wells 3 and 4 the shallow aquifer is 12 and 31 feet thick, respectively, near its center. Toward the center of the main valley the Horse Springs Coulee aquifer thickens to as much as 55 feet (Russell and Eddy, 1971, p. 12).

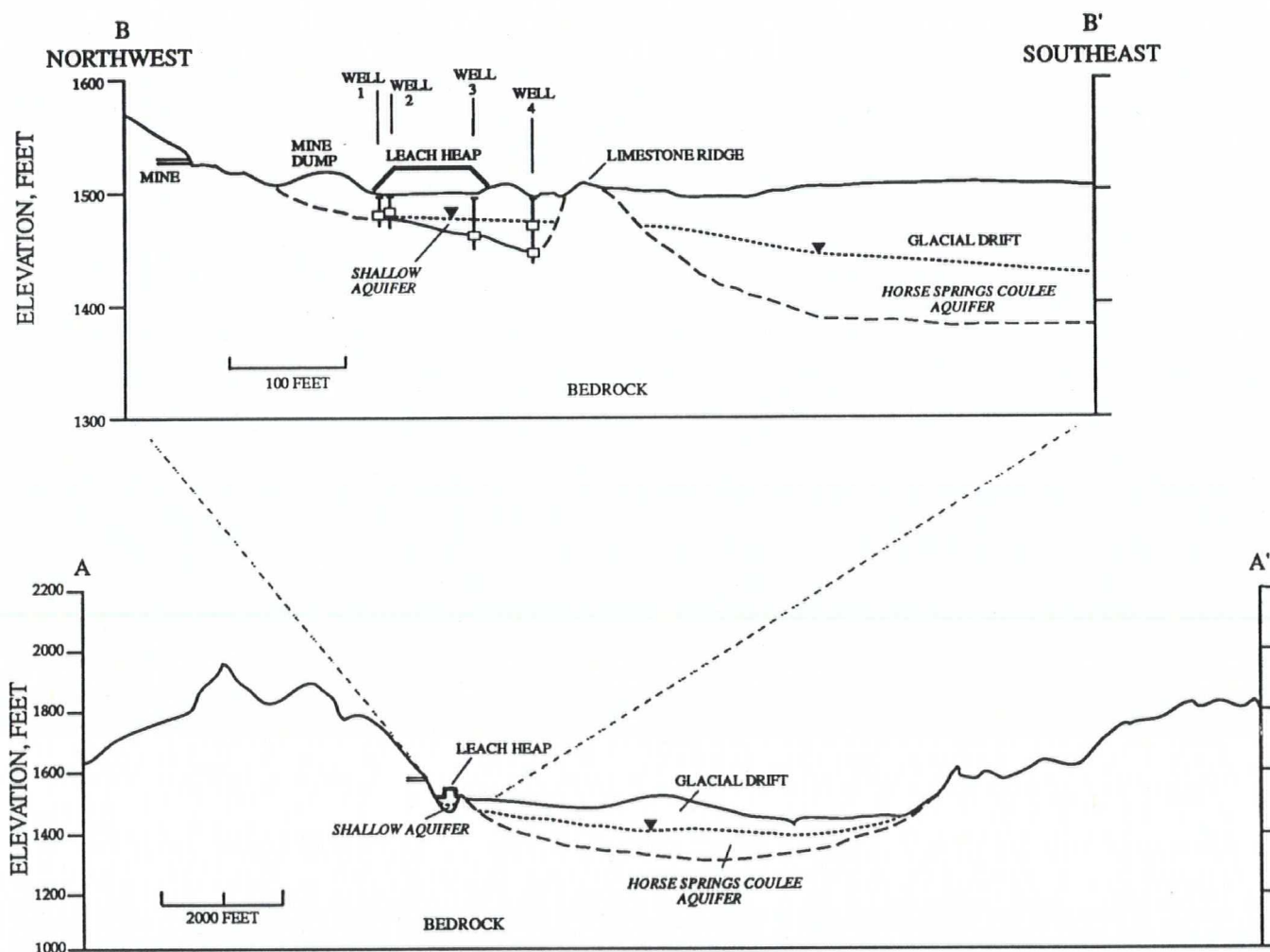


Figure 3.1. Cross sections of Horse Springs Coulee (A-A') and the Silver Mountain Mine site (B-B'). The mine site lies along the west margin of the coulee in an area where the main part of the Horse Springs Coulee aquifer is probably separated from the shallow aquifer directly beneath the leach heap by a bedrock ridge of limestone. The location of the cross section is noted on figure 2.4. The monitoring wells are projected onto the upper cross section.

3.5.2.2 Static Water Levels

The static water levels (SWL) in feet below ground surface are listed for each of the monitoring wells in Table 3.3 and are shown plotted with respect to time in Figure 3.2. Figure 3.2 also shows a three-point moving average of the measurements to minimize the effect of short-term transient events.

The time-averaged water levels for Wells 1 and 2 show rising water level from December through March and dropping level from April through June. This trend indicates a seasonal response that reflects higher precipitation in the winter and higher evapotranspiration and lower precipitation, both of which lead to less aquifer recharge, during the late spring and summer months.

Well 3 also shows a rise in water level through the spring, but a less marked drop during summer. Seasonal changes in vertical recharge at Well 3 may be delayed or more variable in comparison to Wells 1 and 2 because of the well's distance from the recharge area to the west, the greater thickness of the aquifer at Well 3, or a finer-grained lithology occurring at its screened interval.

Though fewer SWL measurements were obtained for Well 4, the results are consistent with those in Wells 1 and 2.

3.5.2.3 Hydraulic Conductivity and Flow Velocity

The hydraulic conductivity of the shallow aquifer was estimated by means of a slug test (Section 2.8.2.2). The hydraulic conductivity was calculated as volume per day per cross-sectional area of aquifer, (ft^3 per day per ft^2 , or simply feet per day). Based on the slug test data (provided in Appendix C), the hydraulic conductivity, K , was determined to be 1.99×10^{-2} feet per day ($7 \times 10^{-6} \text{ cm/s}$).

The velocity of the ground water flow is based on the following equation: $v = [K(h_1 - h_2)]/nL$. K , the hydraulic conductivity, is determined above. The hydraulic gradient, $(h_2 - h_1)/L$ (ie. the difference in hydraulic heads divided by the hydraulic length between wells), is based on the water levels of the monitoring wells. The porosity, n , is expressed as a decimal (0.175) and is an average of the range of porosities for glacial drift (0.1-0.25).

Using these parameters, the ground water velocity was estimated at 1.3 feet/year. Such a velocity would require about 10 months for groundwater to travel one foot. This relatively low velocity should be representative of the fine-grained parts of the shallow aquifer. Velocities are expected to be higher where the shallow aquifer lacks fine-grained material.

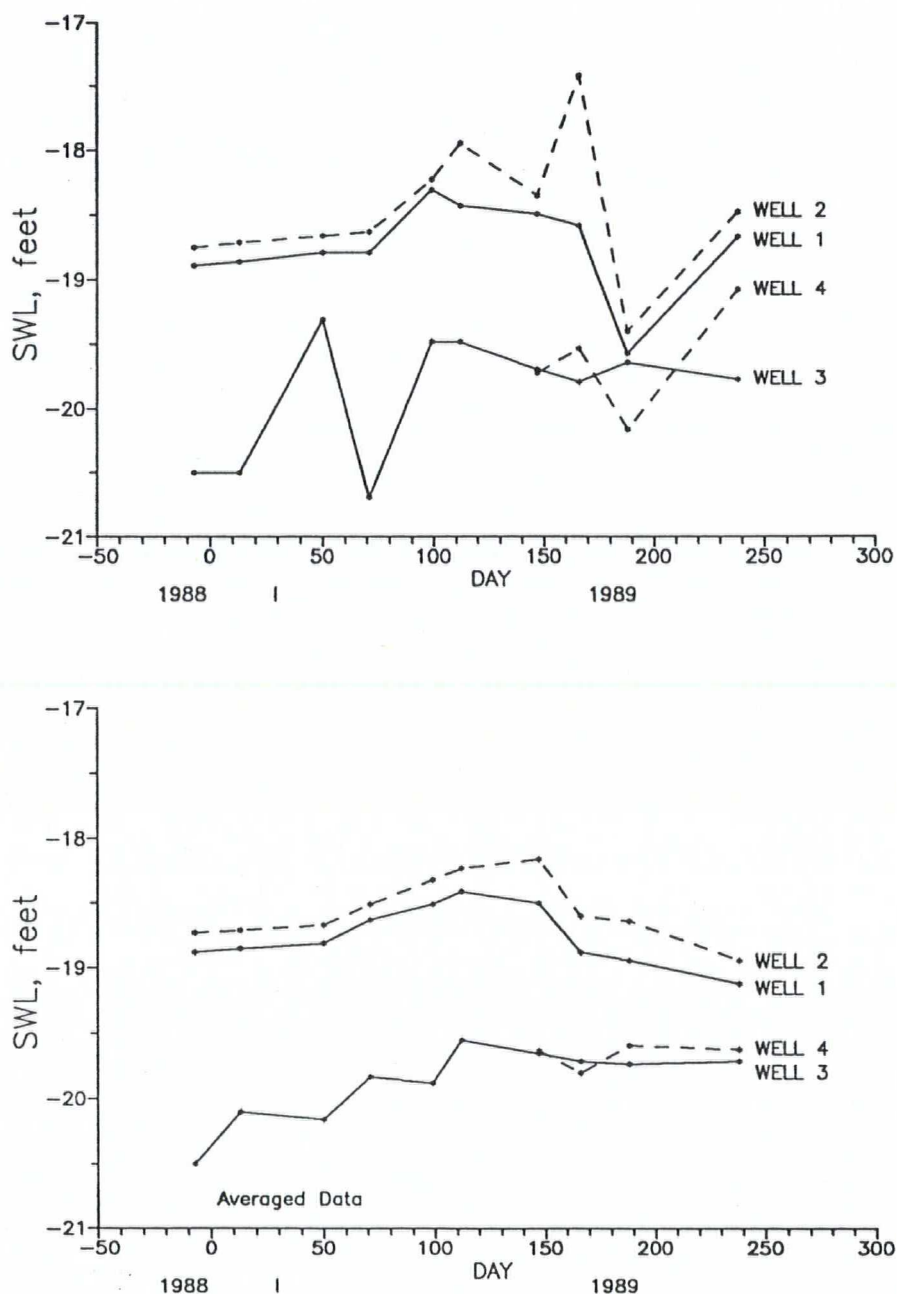


Figure 3.2. The trend in static water level (SWL) from December 1988 through July 1989 in monitoring wells. The distance from the ground surface to the water table increases toward the bottom of each graph. The top graph displays raw data; the bottom graph displays a three-point moving average of the data. The bottom graph shows the seasonal variation of rising water level during the spring and falling water level during summer 1989.

Table 3.3 List of results for field parameters.

SWL-static water level in feet, TEMP-temperature in oC,
EC-electrical conductivity in uS, EH-redox potential in mv.

DATE	SWL	MONITORING WELL 1				STOCK TANK			
		TEMP	EC	PH	EH	TEMP	EC	PH	EH
Nov 23, 1988	18.89		488	7.65	205				
Dec 13, 1988	18.86	11.6	1140	7.09	263				
Jan 19, 1989	18.79	9.7	1190	7.38	253				
Feb 9, 1989	18.79	10.5	1160	7.89	276				
Mar 9, 1989	18.31	10.8	921	8.03	185				
Mar 22, 1989	18.43	11.5	902	7.94	174				
Apr 26, 1989	18.49	11.2	912	7.39	317	13.2	873	8.24	210
May 15, 1989	18.58	11.6	876	7.82	286	11.7	845	8.19	245
Jun 6, 1989	19.57	11.7	896	7.9	321	14.2	855	8.46	256
Jul 26, 1989	18.67	11.7	975	7.45	98	15.0	827	8.45	160
DATE	SWL	MONITORING WELL 2				SEEP			
		TEMP	EC	PH	EH	TEMP	EC	PH	EH
Nov 23, 1988	18.75		1070	7.72	206				
Dec 13, 1988	18.71	10.6	1108	7.34	97				
Jan 19, 1989	18.66	9.5	1210	7.6	276				
Feb 9, 1989	18.63	10.3	1173	7.97	276				
Mar 9, 1989	18.23	11.2	1163	8.19	194				
Mar 22, 1989	17.95	11.0	1112	8.05	239				
Apr 26, 1989	18.35	10.4	1194	8.13	268	15.9	178	7.99	199
May 15, 1989	17.42	10.8	1163	7.97	305	20.4	966	7.77	254
Jun 6, 1989	19.40	10.8	1112	8.09	245	29.7	991	7.60	-21
Jul 26, 1989	18.48	11.4	1144	7.82	148	21.0	970	7.87	148
DATE	SWL	MONITORING WELL 3				IRRIGATION 1			
		TEMP	EC	PH	EH	TEMP	EC	PH	EH
Nov 23, 1988	20.50		550	8.13	181				
Dec 13, 1988	20.50	10.6	629	7.71	283				
Jan 19, 1989	19.31	10.8	635	8	225				
Feb 9, 1989	20.69	11.3	574	7.98	248				
Mar 9, 1989	19.48	11.2	597	7.75	189				
Mar 22, 1989	19.48	11.6	709	7.39	225				
Apr 26, 1989	19.69	11.0	739	8.12	226				
May 15, 1989	19.79	11.7	1582	7.26	309	12.8	590	6.95	248
Jun 6, 1989	19.64	11.7	1662	7.25	225				
Jul 26, 1989	19.77	11.6	1373	7.69	216	12.6	629	7.6	218
DATE	SWL	MONITORING WELL 4				IRRIGATION 2			
		TEMP	EC	PH	EH	TEMP	EC	PH	EH
Apr 26, 1989	19.72								
May 15, 1989	19.53	11.7	560	8.08	271	12.7	594	7.43	249
Jun 6, 1989	20.16	12.0	616	8.16	303	12.9	515	7.62	30
Jul 26, 1989	19.08	12.0	478	7.92	202	12.8	549	7.53	266
DATE	SWL					RESIDENCE			
		TEMP	EC	PH	EH	TEMP	EC	PH	EH
Jun 6, 1989						13.3	553	7.56	10
Jul 26, 1989						12.7	613	7.44	204

3.5.2.4 Flow Direction

After the installation of the four monitoring wells, a detailed ground survey was performed. The survey, described in Section 2.1, provided the relative elevations of the well heads, as well as azimuth angles and distances between them. Based on this information and periodic measurements of static water levels at the wells, the direction of ground water flow was calculated. An average flow direction of S 67.3° E was obtained for measurements from April to June 1989.

Based on the four full sets of static water level data from April, May, June, and July, water table surfaces were drawn which indicate flow direction toward the southeast. Figures 3.3-3.6 show the water table contours obtained at the site. Flow direction runs at right angles to the contour lines and is indicated by small arrows.

3.5.2.5 Ground Water Budget

A ground water budget was calculated to estimate changes in ground water storage due to inflow to and outflow from the shallow aquifer. If inflow equals outflow, then there are no changes in the volume of the water in storage, and the aquifer should be in a steady state condition. The shallow aquifer considered in this budget calculation is defined by the area between the western outcrop of the anarchist group and the limestone ridge east of the site. This area includes the leach heap and mine dumps, where spills and contaminated runoff may have entered the ground water.

Inflow:

Because the horizontal ground water velocity calculated in Section 3.6.1.3 is very slow, horizontal inputs to the shallow aquifer may be neglected in the aquifer storage calculations. The aquifer is largely recharged by the infiltration of water through the overlying soil. Recharge occurs during the spring and fall months when the ground is not frozen, less evaporation is occurring, and precipitation or snowmelt is present. In spring, there is very little overland flow at the site. The water from melting snow tends to infiltrate the soil immediately or pool briefly with some evaporation occurring, followed by infiltration. As a result, much of the water from winter precipitation probably enters the aquifer. On the basis of these observations, a 30% recharge rate has been used in calculating the aquifer storage (i.e. taking evaporation into account, 30% of the total precipitation on the site is assumed to enter the aquifer below). Assuming no horizontal inflow and yearly

rainfall and snowmelt of 15 inches, a recharge rate of about 122,000 gallons per acre per year is estimated, using the following equation:

$$\text{Recharge} = (1 \text{ acre}) (43,560 \text{ ft}^2/\text{acre}) (30\%) (15 \text{ in of rain/yr}) (7.48 \text{ gal/ft}^3) (1 \text{ ft}/12 \text{ in}) = 122,000 \text{ gallons/acre-year}$$

Outflow:

The nearest domestic and irrigation wells, described in Section 2.8.1, are several miles away and should not affect on-site hydrology. Both the seep and the water in the stock tank from the mine drainage are associated with discharge from bedrock rather than the glacial drift which makes up the shallow aquifer. As with inflow, horizontal flow is not considered a major path of outflow since the calculated flow velocity is extremely low (see flow velocity, Section 3.6.1.3). Thus, the main ground water discharge from the shallow aquifer occurring at the site is assumed to flow downward to fractured zones in the bedrock. Since no ground water discharges to the surface from the shallow aquifer, outflow must equal the inflow minus any change in storage that may occur.

Ground Water Storage:

The static water level measurements taken at the monitoring wells and plotted for a seven month period in Figure 3.3 indicate that the water level change was never less than 10% of the aquifer thickness at the thickest part of the aquifer (Wells 3 and 4). Although a small seasonal change in storage occurs, the annual change in storage is estimated to be negligible and the aquifer may be assumed to be in a steady state. Therefore, outflow from the shallow aquifer should equal inflow, or 122,000 gallons/year.

3.5.2.6 Ground Water Temperature

Monthly temperature measurements are listed in Table 3.3 and are plotted for the period between December 1988 and June 1989 in Figure 3.8. Well 4 temperatures were only measured twice, due to the later installation of the well, and appear comparable to temperatures for Well 3. In general, ground water at wells 1 and 2 is cooler during the winter months and warmer in spring and summer where the aquifer is closest to the surface. These wells are located in a shallow part of the aquifer near the bedrock outcrop to the west and show greater seasonal water temperature changes than Well 3, located further east, toward the thickest part of the shallow aquifer.

GROUND WATER SURFACE 4/26/89

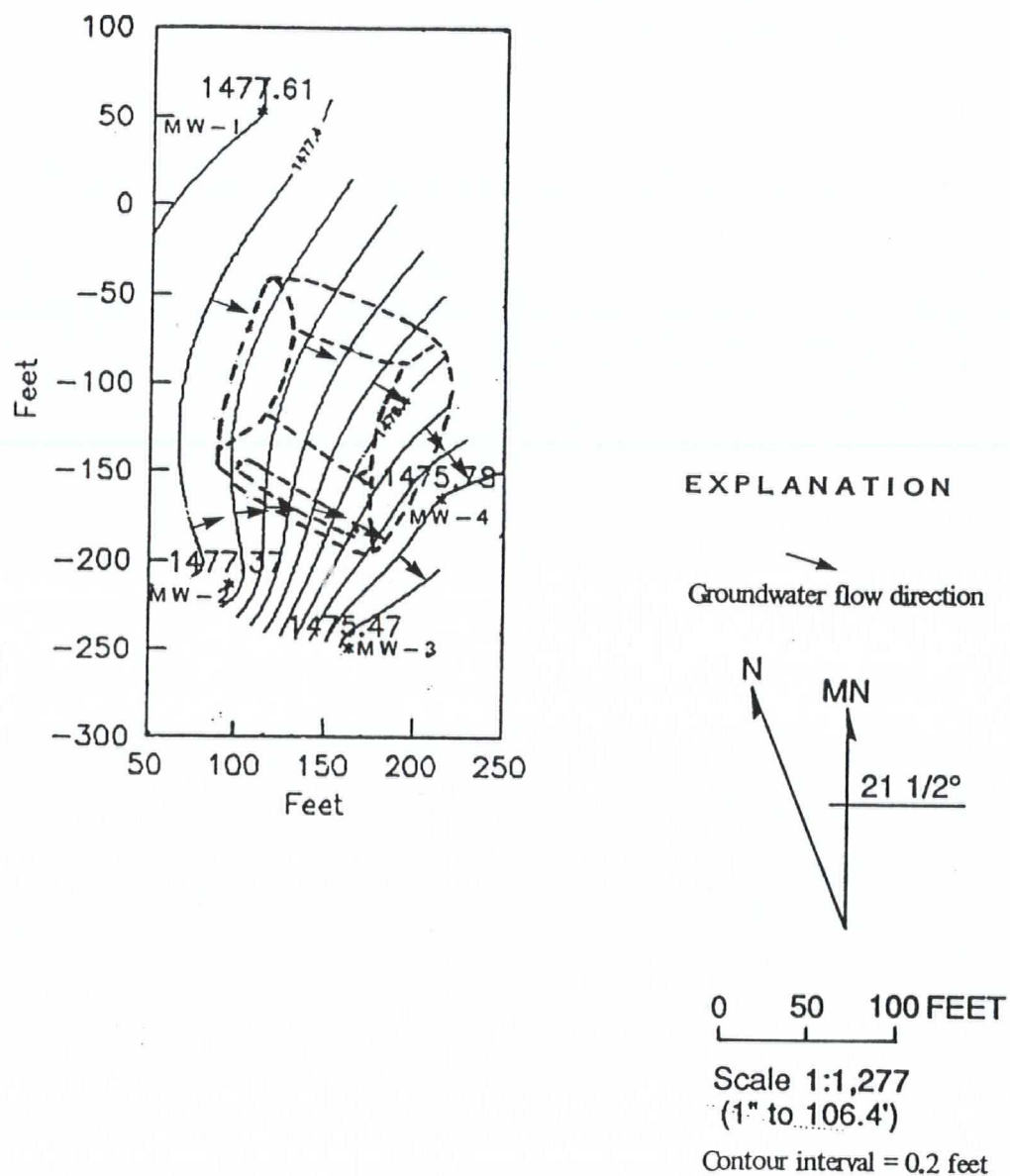
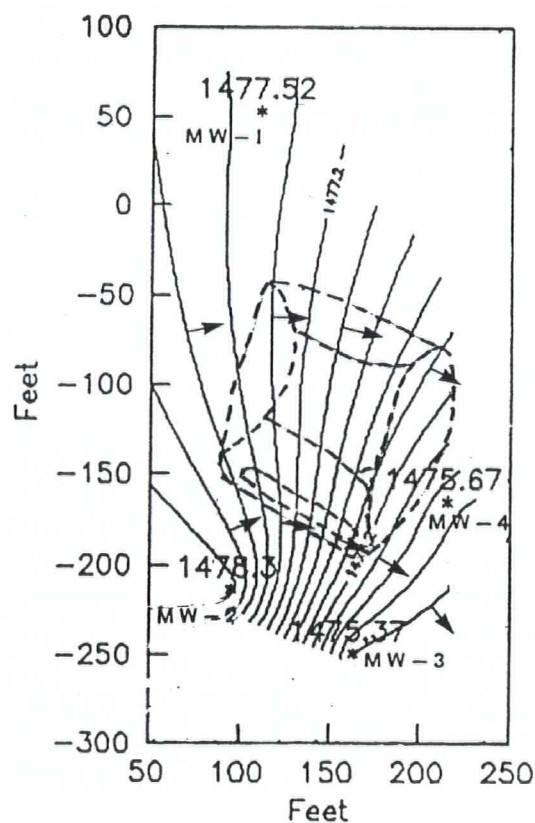


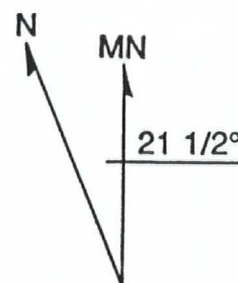
Figure 3.3. Ground water surface, April 1989.

GROUND WATER SURFACE 5/15/89



EXPLANATION

→
Groundwater flow direction



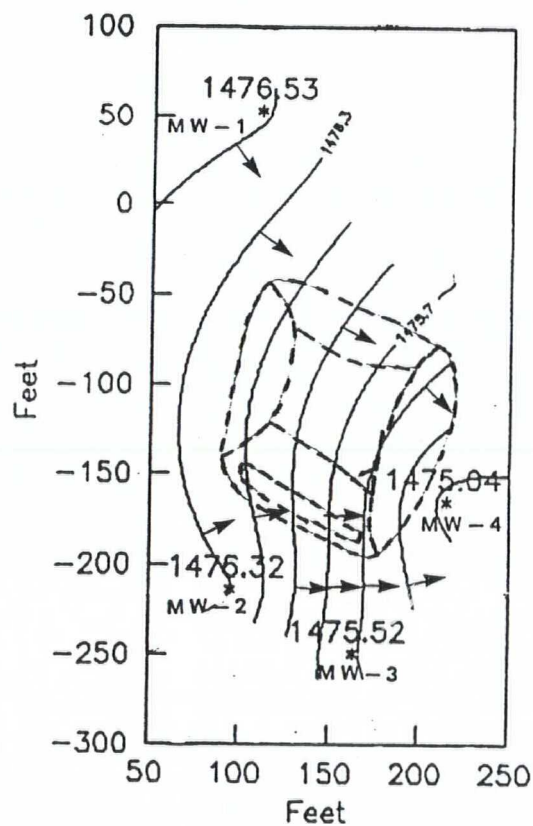
0 50 100 FEET

Scale 1:1,277
(1" to 106.4')

Contour interval = 0.2 feet

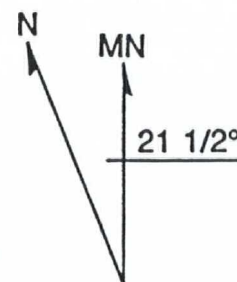
Figure 3.4. Ground water surface, May 1989.

GROUND WATER SURFACE 6/6/89



EXPLANATION

Groundwater flow direction



0 50 100 FEET

Scale 1:1,277
(1" to 106.4')

Contour interval = 0.2 feet

Figure 3.5. Ground water surface, June 1989.

GROUND WATER SURFACE 7/26/89

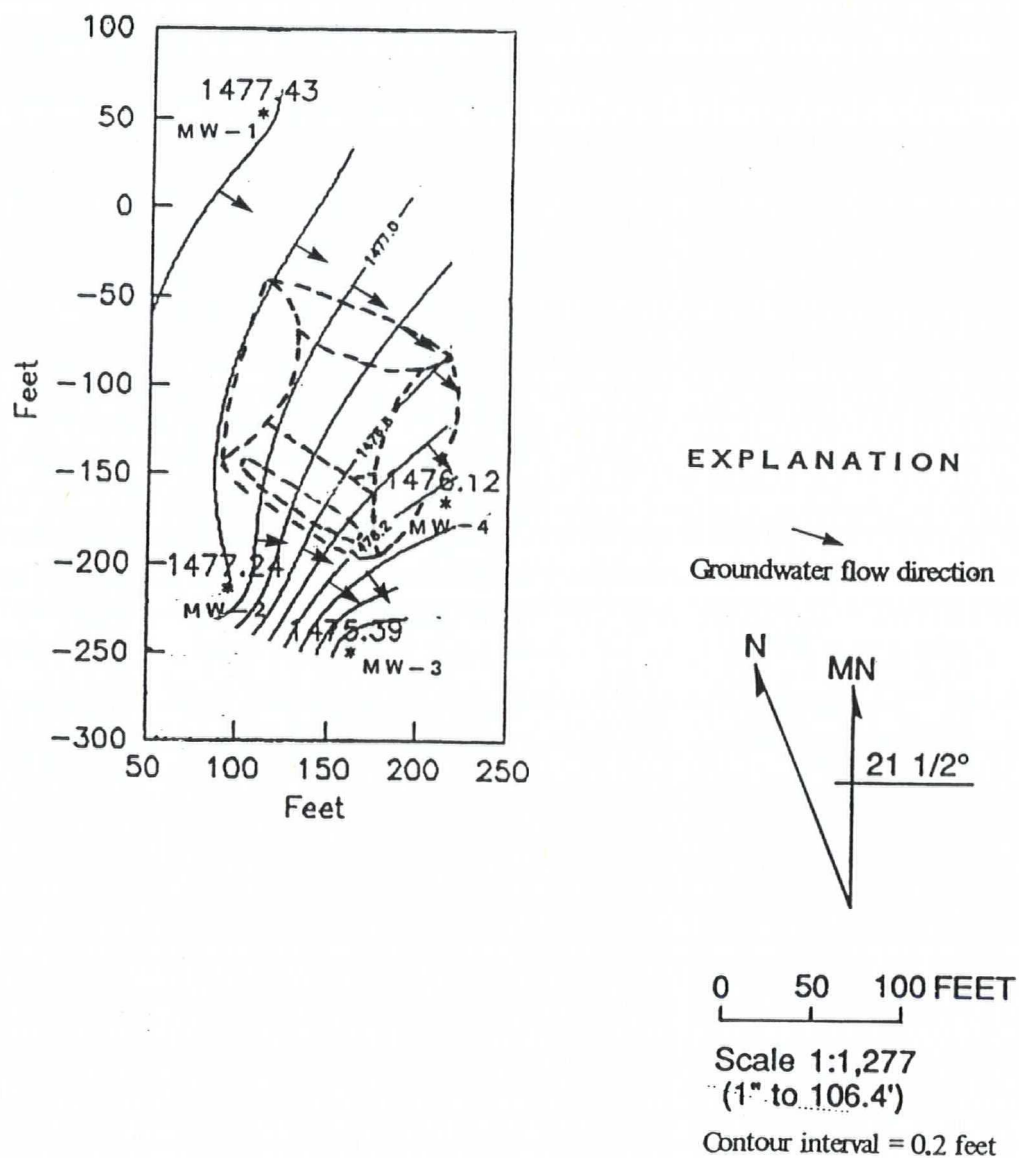


Figure 3.6. Ground Water Surface, July 1989.

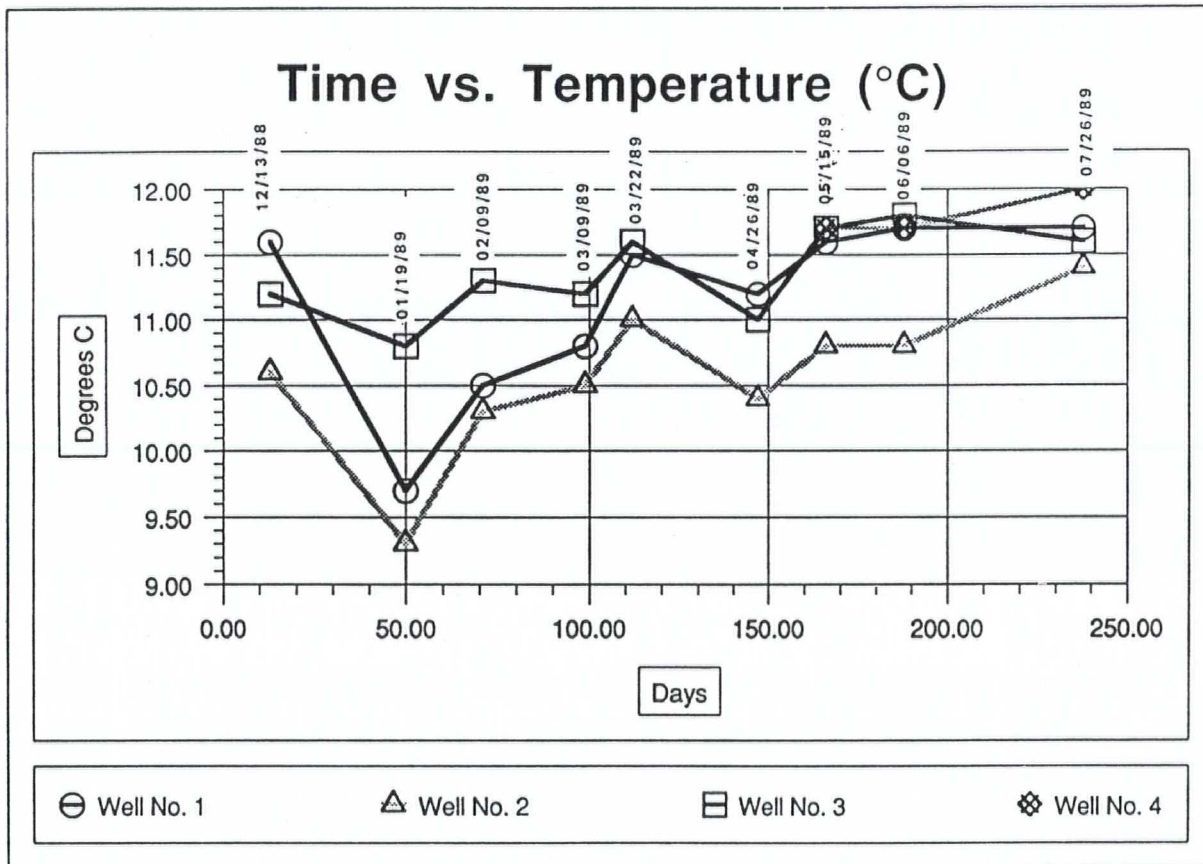


Figure 3.7. Monitoring Well Temperatures.

3.5.3 Field Results

The field analyses for water described in Section 2.9.4 include determinations of pH, Eh, electrical conductivity, and free cyanide concentrations. To assure accuracy and to develop a range of data, field parameters were measured monthly between December 1988 and June 1989. The field results are discussed in this section and presented in Table 3.3. Results of laboratory analyses for other parameters are discussed in Chapter 4.

pH:

The pH in ground water samples from the four monitoring wells, offsite wells, and surface water ranged from 7.0 to 8.5 (Table 3.3). The range of pH values and a six-month average pH obtained for the monitoring wells are as follows:

<u>Well No.</u>	<u>pH range</u>	<u>pH average</u>
1	7.1 - 8.0	7.7
2	7.3 - 8.2	7.9
3	7.3 - 8.1	7.7
4	7.9 - 8.2	8.1

The average pH of the onsite ground water samples was found to be slightly alkaline at pH 7.9. The pH values for offsite wells ranged from 7.0 to 8.0, while site surface waters ranged from 7.9 to 8.5 (Table 3.3).

Eh:

The Eh, or oxidation reduction (redox) potential was measured to provide a rough indication of the preferred oxidation state of dissolved chemical species. The Eh values are expressed in millivolts (mv), with a higher value indicating higher oxidation potential. The range of Eh values obtained at the monitoring wells (Table 3.3) and the six month averages are as follows:

<u>Well No.</u>	<u>Eh Range (mv)</u>	<u>Eh Average (mv)</u>
1.....	144 - 321	259
2.....	97 - 305	238
3.....	189 - 309	240
4.....	271 - 300	287

These Eh values are observed values relative to a silver-silver chloride reference electrode. These values should be

adjusted by about +246 mv to obtain the theoretical potential relative to a Standard Hydrogen Electrode at the temperature of ground water. Eh values obtained for ground water are consistent with shallow ground water in contact with an unsaturated zone. The Eh may be attributed in part to chemical reactions as the water percolates through soil and rock during infiltration to the aquifer and in part to contact with the atmosphere during measurement. Eh may also be affected by the proximity of the monitoring wells to metallic ores at the mine site. Over time the reaction of ground water with metallic sulfides would tend to lower the Eh in water in contact with mineralized rock.

Eh values ranged from 10 to 266 mv for offsite wells and from -21 to 256 mv for site surface waters (Table 3.3).

Electrical Conductivity:

Electrical conductivity was measured in all samples to provide a rough indication of the amount of dissolved solids in water. Conductivity is expressed in microsiemens (μS), the metric equivalent of micromhos (μmho). For high sulfate water such as the ground water at the site, a factor of 0.75 (Hem, 1970) can be used to estimate total dissolved solids in milligrams per liter. Using this factor, the estimated range of total dissolved solids for the site ground water is from 400 to 900 mg/L.

The ranges and average values obtained for electrical conductivity at each monitoring well are as follows:

Well No.	μS range	μS avg
1.....	902 - 1194	1064
2.....	1108 - 1210	1153
3.....	574 - 1662	905
4.....	478 - 616	588

The average of the electrical conductivity for Wells 1 and 2 is 67% higher than the corresponding average for Wells 3 and 4. This large difference may be due to the greater thickness of the shallow aquifer at Wells 3 and 4 than at Wells 1 and 2 and to the greater distance of 3 and 4 from the bedrock recharge area to the west. Where the shallow aquifer is less thick, the smaller water volume is preferentially influenced by contact with the underlying bedrock of Anarchist metasediments. If these metasediments contain a greater proportion of more soluble minerals than the glacial drift, more dissolved solids per volume of water may result in areas where the shallow aquifer receives recharge from adjacent bedrock (Hem, 1970, p. 300).

The distribution pattern of electrical conductivity values appears to vary with time. Examination of Table 3.3 shows that during the period of sample collection from May to July 1989, electrical conductivity clearly increased in a downgradient direction as ground water passed beneath the leach heap. As discussed in Chapter 4 this pattern is consistent with the distribution of other contaminants attributed to the leach heap during the same time period.

Conductivity values ranged from 515 to 629 μS for off-site wells and 178 to 991 μS for on-site surface waters (Table 3.3).

3.5.4 Shallow Aquifer Summary

Ground water recharge of the shallow aquifer primarily occurs during the spring and fall, due to increased precipitation and lower evapotranspiration rates. Approximately 122,000 gallons per acre are recharged each year. Since there is virtually no surface discharge, the aquifer is considered to be in "steady state" or equilibrium condition over the long term.

The pH of the site ground water averages 7.9, which is slightly alkaline. The ground water contains a large amount of dissolved solids, ranging between 400 and 900 mg/L, based on electrical conductance.

The well logs for the monitoring wells show a maximum aquifer thickness of 31 feet, with a thickening trend toward the east. Ground water flows toward the southeast to south, and recharge occurs west of the site, where the aquifer abuts the metasedimentary bedrock. Downgradient from the site, ground water probably joins the main part of Horse Springs Coulee-Aeneas Lake aquifer, which flows south toward Aeneas Lake. The extremely slow flow rate of the shallow aquifer beneath the site suggests that any contamination to reach the ground water would probably remain localized for some time.

3.6 ECOLOGY

Like many areas of Washington east of the Cascade Mountains, the Silver Mountain Mine site is part of a sagebrush ecosystem. This type of ecosystem has been studied extensively, particularly in the area around Hanford reservation, and is characterized by a predominance of sagebrush and bunchgrass.

A detailed description of the ecosystem is provided in Chapter 7. The general plant community and wildlife in the area are discussed and site specific information is used to identify ecological endpoints of concern.

REFERENCES

- Ecology and Environment, Inc., 1985, Preliminary site inspection report of Silver Mountain Mine, Loomis, Washington: Ecology and Environment, Inc., Seattle, Washington, 13 p.
- Hem, J.D., 1970, Study and interpretation of the chemical characteristics of natural water: U.S. Geological Survey Bulletin 1473, 363 p.
- Rinehart, C.D. and Fox, K.F., Jr., 1972, Geology and mineral deposits of the Loomis quadrangle, Okanogan County, Washington: Washington Division of Mines and Geology Bulletin 64, 124 p.
- Rinehart, C.D. and Fox, K.F., Jr., 1976, Bedrock geology of the Conconully quadrangle, Okanogan County, Washington: U.S. Geological Survey Bulletin 1402, 58 p.
- Russell, R.H. and Eddy, P.A., 1972, Geohydrologic evaluation of Aeneas Lake-Horse Springs Coulee, Okanogan County, Washington: Washington Department of Ecology, Office of Technical Services Investigations, 29 p.

CHAPTER 4 - NATURE AND EXTENT OF CONTAMINATION

This chapter discusses the distribution of contaminants at Silver Mountain Mine in the context of the chemical composition of four features of the site: the mined materials in the leach heap and mine dump, the soils adjacent to the piles and underlying the leachate pond, the ground water, and surface water.

4.1 LEACH HEAP AND MINE DUMP

4.1.1 Location of Sample Sites

Figure 2.3a shows the location of rock samples taken from the heap-leach pile and the mine dump. Twenty-six samples were collected to characterize the inorganic composition of the heap and dump, and eight samples were collected to determine the hazardous waste leaching characteristics of the heap material. Sample locations were selected to provide coverage of three areas of the heap: 10 samples from the top, six from the north and east edges, and five at the base of the south side of the heap. Three samples were collected from the mine dump. A discussion of the sampling procedures is provided in Section 2.4 and sample lithology is listed on Table 3.1.

4.1.2 Analytical Results

Samples of heap and mine dump material were analyzed for the inorganic parameters listed in Table 2.2. Results are shown in Table 4.1 and 4.2, where the first six digits of the sample numbers refer to the date of sampling and the last two digits correspond to the sample site number shown in Figure 2.3.

The concentrations of metals listed in the Table 4.1 are based on acid-soluble analytical procedures (ICP and GFAA). The results are consistent with the types of rock identified during the field investigation, including sulfide-bearing metaphyllite, greenschist, and vein quartz (Table 3.1).

Major chemical constituents that generally occur at concentrations exceeding 10,000 mg/kg (1%) include calcium, aluminum, and magnesium in order of generally decreasing abundance. Minor constituents, ranging from 1000 to 10,000 mg/kg, include sodium, potassium, manganese, and arsenic. Several trace elements occur at concentrations of less than 1000 mg/kg. Of these, the most abundant are zinc at concentrations up

to 561 mg/kg, copper to 546 mg/kg, lead to 267 mg/kg, barium to 110 mg/kg, nickel to 51 mg/kg, and silver to 39 mg/kg. Mercury ranges from values below the detection limit of 0.02 mg/kg to maximum concentrations of 0.36 mg/kg in the heap and 0.78 mg/kg in the mine dump.

Cyanide results are shown in Table 4.1 both as total cyanide and as weak acid dissociable (WAD) cyanide. Total cyanide ranges up to 173 mg/kg in the heap with a corresponding weak acid dissociable cyanide value of 15.1 mg/kg.

Examination of Table 4.1 and Figure 4.1 indicates that preferential concentration of cyanide occurs at the toe of the heap, with values about ten times those occurring elsewhere in the heap. Additionally, a few other elements including sodium, manganese, copper, zinc, and lead, are concentrated along with cyanide at the toe of the heap, whereas chromium appears to be depleted from the top of the heap.

Arsenic occurs at moderate to high concentrations in both the mine dump and the leach heap, as shown in Figure 4.2. The three highest values of arsenic are 652 mg/kg at the toe of the heap, 626 mg/kg on top of the heap and 1075 mg/kg in the mine dump.

The results of toxicity characteristic testing of eight samples are listed in Table 4.2, four samples as EP-Toxicity (extraction procedure toxicity test) and four as TCLP (toxicity characteristic leaching procedure). The EP toxicity tests show that heap material does not exceed inorganic leachate criteria that would designate the waste as hazardous waste under federal RCRA regulations (40 CFR Section 261) or dangerous waste under Washington State Dangerous Waste regulations (WAC 173-303).

Of the eight parameters analyzed in the toxicity characteristic tests, lead, cadmium, and barium appear to leach from the heap material at concentrations greater than that found in ground water samples. The test procedures are performed under acidic conditions and would indicate that these metals should be relatively high in leachate if the heap were leached under acidic conditions. Both the heap and ground water are slightly alkaline, however, and acidic leaching should not occur under current conditions. Additional discussion of the leaching characteristics of heap material with respect to cyanide and arsenic is included in Chapter 5.

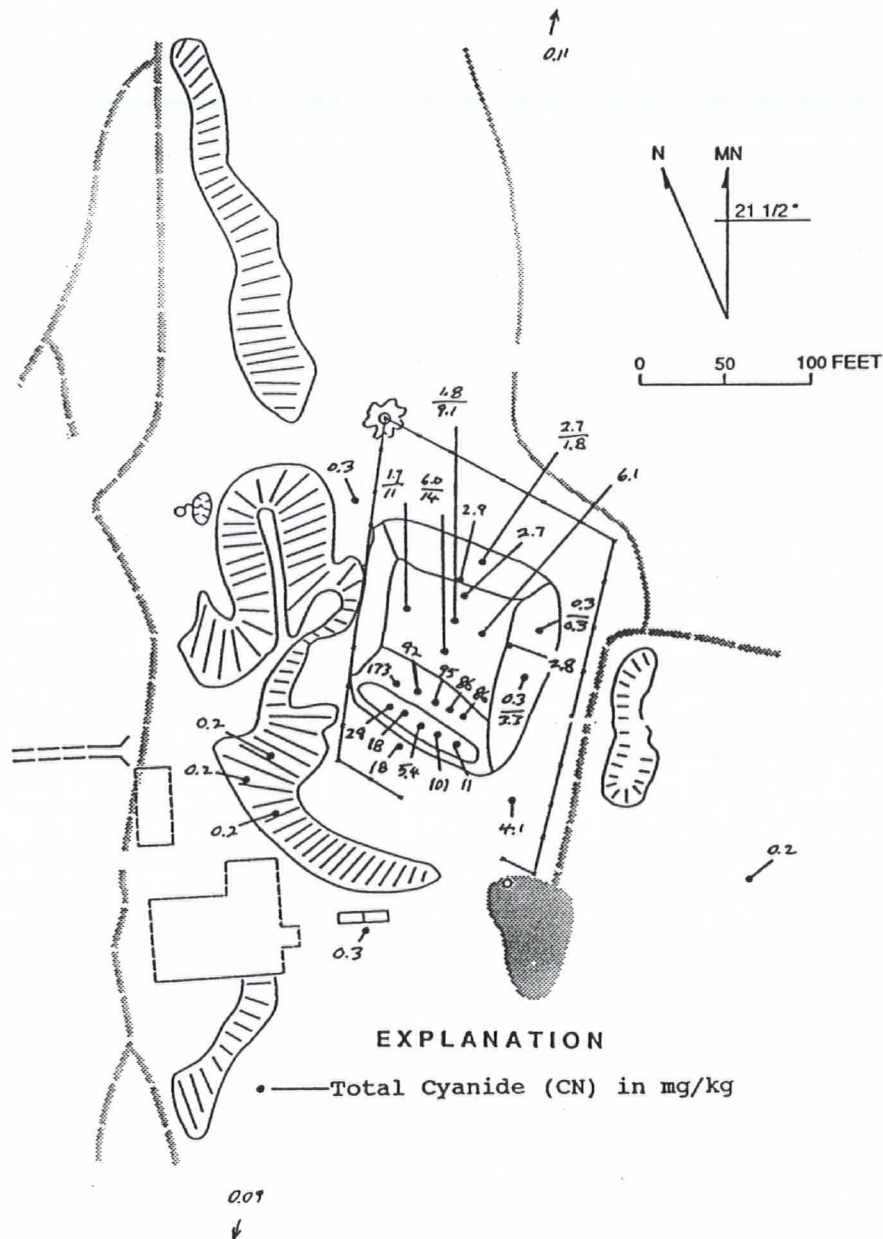


Figure 4.1. The distribution of total cyanide in the leach heap, mine dump, and soil. Sample locations are numbered in figure 2.3a. Data are in mg/kg. The two values noted by arrows are background samples no. 53 and 54. Values separated by horizontal line represent upper and lower samples at the same location.

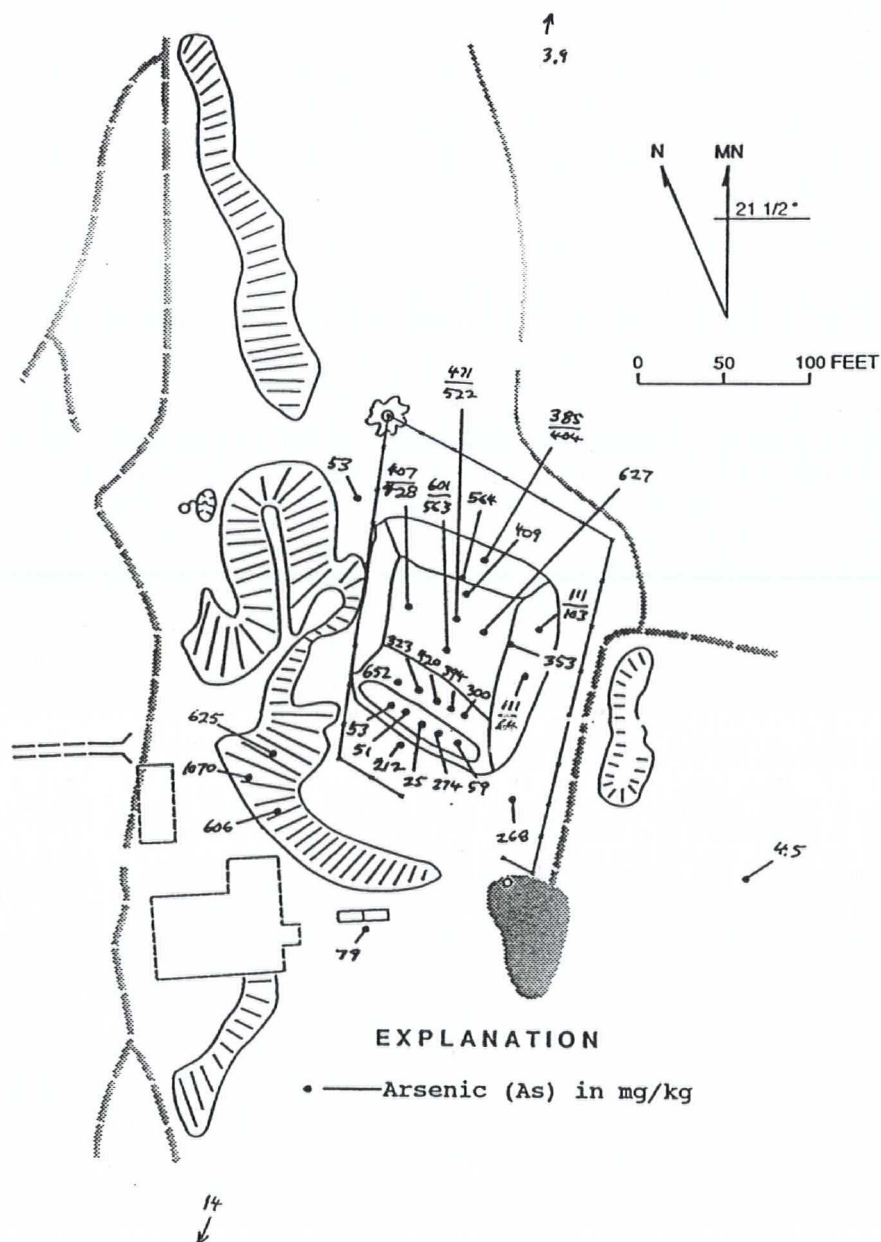


Figure 4.2. The distribution of arsenic in the leach heap, mine dump, and soil. Sample locations are numbered in figure 2.3a. Data are in mg/kg. The two values noted by arrows are background samples no. 53 and 54. Values separated by horizontal line represent upper and lower samples at the same location.

RI CHAPTER 4
PAGE 5

Table 4.1. Inorganic analytical results for leach heap, mine dump, and soils. Metals and Cyanide

Location	WFOC #	Al	Sb	As	Ba	Be	Cd	Ca	Cr	Co	Cu	Fe	Pb	Mg
Beneath pond	890330005	10400	0.43 J	58.6	43.5	0.5 UJ	1.4 J	22700	11.6	14.6 J	78.8	38900 R	22.6 J	8330
Beneath pond	890330006	8890	0.48 J	274	59.5	0.5 UJ	6.8 J	77000	12.9	11.1 J	504	43000 R	43.6 J	11300
Beneath pond	890330007	12900	0.44 J	25.1	113	0.5 UJ	1.7 J	5160	13.0	11.0 J	67.1	19100 R	23.3 J	3740
Beneath pond	890330008	12000	0.44 J	50.7	80.1	0.5 UJ	1.2 J	15100	15.5	10.7 J	98.7	31300 R	21.8 J	6450
Beneath pond	890330009	11300	1.3 J	52.6	92.3	0.5 UJ	1.2 J	8040	13.2	9.4 J	97.0	29200 R	24.3 J	4540
Beneath pond	890330009D	11100		48.4	94.8	0.5	1.3	6690	12.5	9.3	96.9	28700	19.0	4240
Top of heap	890330010	5910	1.8 J	428	50.5	0.5 UJ	1.6 J	8150	5.5	7.0 J	43.9	21400 R	105 J	4730
Top of heap	890330011	8300	2.8 J	407	57.5	0.5 UJ	1.5 J	8880	9.6	11.0 J	49.5	26900 R	171 J	4700
Top of heap	890330012	6850	1.3 J	522	55.1	0.5 UJ	2.8 J	10900	8.6	8.7 J	71.1	25300 R	63.3 J	5000
Top of heap	890330013	5910	1.6 J	471	33.7	0.5 UJ	2.1 J	10300	8.5	9.4 J	61.8	24700 R	112 J	5140
Top of heap	890330014	5610	3.0 J	563	38.0	0.5 UJ	2.0 J	12100	5.8	9.3 J	46.2	27400 R	142 J	4820
Top of heap	890330015	5130	1.0 J	601	34.4	0.5 UJ	2.7 J	9870	7.3	9.8 J	56.2	33500 R	112 J	4330
Top of heap	890330016	4410	1.2 J	627	30.3	0.5 UJ	1.7 J	12000	3.3	9.0 J	46.7	26900 R	77.0 J	4420
Top of heap	890330017	5720	2.5 J	353	45.9	0.5 UJ	1.9 J	9180	6.9	9.1 J	48.7	26800 R	77.2 J	4130
Top of heap	890330018	5940	7.9 J	564	52.3	0.5 UJ	4.8 J	9060	6.8	8.5 J	65.8	27400 R	75.2 J	4410
Top of heap	890330019	6400	1.5 J	409	59.2	0.5 UJ	1.3 J	10400	6.1	8.8 J	29.8	24700 R	77.1 J	4300
S. Toe of heap	890331020	5360	0.30 J	300	29.9	0.5 UJ	2.4 J	23300	8.9	8.5 J	535	42400 R	103 J	3720
S. Toe of heap	890331021	8000	0.56 J	323	42.4	0.5 UJ	3.2 J	19500	14.5	9.9 J	338	40900 R	112 J	4830
S. Toe of heap	890331022	6530	0.27 J	394	32.1	0.5 UJ	4.0 J	27900	11.3	8.5 J	546	46300 R	180 J	3910
S. Toe of heap	890331023	6090	0.61 J	420	25.5 J	0.5 UJ	2.6 J	24500 J	15.4 J	8.3 J	500 J	41300 R	267 J	3710 J
S. Toe of heap	890331024	5430	0.74 J	652	35.2 J	0.5 UJ	3.1 J	18000 J	10.5 J	8.7 J	359 J	30500 R	189 J	3940 J
S. Toe of heap	890331024D	5830	0.91 J	614	40.4	0.4	3.9	18800	9.1	8.8	348	30800	212	4100
N. and E. edge	890331025	12300	0.91 J	385	61.7 J	0.5 UJ	1.4 J	12700 J	16.1 J	16.4 J	76.3 J	37500 R	56.9 J	8920 J
N. and E. edge	890331026	12000	0.71 J	412	52.2 J	0.5 UJ	1.8 J	12600 J	16.5 J	16.9 J	79.2 J	38300 R	47.6 J	8820 J
N. and E. edge	890331027	12000	0.60 J	404	50.4 J	0.6 J	1.7 J	13400 J	16.0 J	18.4 J	200 J	42400 R	46.4 J	9170 J
N. and E. edge	890331028	12200	0.34 R	103	55.3 J	0.5 UJ	1.3 J	14200 J	12.9 J	16.1 J	63.8 J	40300 R	29.7 J	7250 J
N. and E. edge	890331029	13400	0.48 R	87.2	66.7 J	0.5 UJ	1.5 J	14100 J	13.5 J	15.7 J	60.4 J	40900 R	15.4 UJ	7840 J
N. and E. edge	890331030	12200	0.61 J	111	46.5 J	0.8 J	1.7 J	24500 J	10.6 J	22.3 J	77.3 J	45600 R	18.4 UJ	11200 J
N. and E. edge	890331031	15100	2.1 J	64.2	110 J	0.6 J	1.0 J	8700 J	14.5 J	12.0 J	67.4 J	26600 R	19.5 J	5930 J
N. and E. edge	890331032	8850	0.86 J	349	36.1 J	0.5 UJ	1.3 J	15100 J	9.2 J	13.7 J	89.9 J	32800 R	42.5 J	7590 J
Mine Dump	890406033	1300	11.3 J	606	28.7 J	0.5 UJ	1.3 J	2520 J	1.0 UJ	1.3 UJ	30.2 J	15800 R	212 J	777 J
Mine Dump	890406034	1800	30.9 J	1080	36.6 J	0.5 UJ	1.4 J	6330 J	1.0 J	2.7 J	40.2 J	17000 R	189 J	822 J
Mine Dump	890406035	7120	2.0 J	625	55.9 J	0.5 UJ	1.7 J	14800 J	11.3 J	14.9 J	48.5 J	38600 R	75.1 J	8880 J
Nearby Soil	890406036	8170	3.7 J	79.3	77.6 J	0.6 J	1.6 J	11500 J	9.4 J	10.5 J	64.0 J	26100 R	35.1 J	5490 J
Nearby Soil	89050437	11200	1.2 J	4.5	109 J	0.5 UJ	0.5 J	2990 J	13.7 J	5.4 J	23.6 J	13100 R	9.3 UJ	4000 J
Nearby Soil	89050438	11100	8.2 J	268	33.7 J	0.7 J	1.6 J	16600 J	13.0 J	16.3 J	72.0 J	38900 R	88.9 J	9420 J
Nearby Soil	89050439	9190	4.8 J	212	35.5 J	0.5 UJ	2.0 J	17800 J	11.7 J	12.4 J	118 J	39200 R	56.1 J	7740 J
Nearby Soil	89050440	11600	4.5 J	53.1	49.5 J	0.5 UJ	1.6 J	13500 J	10.2 J	14.5 J	94.1 J	44200 J	45.9 J	9880 J
Background	890607053	15100	0.38 J	13.9	127	0.5	1.0 UJ	3620	15.1	8.0	28.3	19700	9.2	4230
Background	890607053D	15300		9.8	129	0.4	0.0	3660	14.1	7.9	29.1	18700	4.0	4280
Background	890607054	10400	0.47 J	3.9	92.2	0.5	1.0 UJ	2890	15.9	6.9	22.1	13700	6.4	4320

- Notes: 1. All units in milligrams/kilogram of sample (DRY BASIS).
2. Sample no. suffix "D" = duplicate
3. "----" Denotes sample not analyzed.
4. U denotes compound not detected. Value given is the level of quantification.
5. UJ denotes compound not detected. Value given is the estimated level of quantification.
6. J denotes compound quantification is estimated.
7. R denotes data are unusable. Compound may or may not be present.

RI CHAPTER 4
PAGE 6

Table 4.1. (Continued)

Location	Mn	Hg	Mo	Ni	K	Se	Ag	Na	Tl	V	Zn	CN(WAS)	CN(Total)
Beneath pond	405	0.10 J	1.0 U	31.1	673	0.43 R	4.2 J	275	0.33 R	21.6	120	J 2.4 J	11.3 J
Beneath pond	605	1.5 J	1.0 U	48.2	830	2.4 R	29.9 J	578	0.11 R	19.8	801	J 8.9 J	101 J
Beneath pond	474	0.10 J	1.0 U	25.8	1920	0.23 R	1.9 J	761	0.97 J	24.4	86.4 J	0.4 J	5.4 J
Beneath pond	381	0.23 J	1.0 U	31.4	1260	0.32 R	6.6 J	434	0.25 UJ	27.3	91.2 J	2.6 J	18.2 J
Beneath pond	440	0.45 J	1.0 U	29.5	1160	0.46 R	3.7 J	210	0.15 UJ	22.1	129	J 1.8 J	28.6 J
Beneath pond	428	---	1.0 U	27.5	1240		3.4	213		22.0	129		
Top of heap	532	0.11 J	1.0 U	19.0	876	0.63 R	1.4 J	711	0.24 UJ	9.0	208	J 1.0 J	10.6 J
Top of heap	645	0.07 J	1.0 U	26.2	894	0.74 R	9.0 J	607	0.99 J	13.1	178	J 0.4 J	1.7 J
Top of heap	492	0.11 J	1.0 U	26.7	816	0.54 R	2.9 J	500	0.76 J	14.1	354	J 1.1 J	9.1 J
Top of heap	499	0.09 J	1.0 U	33.8	613	0.43 R	2.7 J	409	0.01 UJ	11.0	229	J 0.3 J	1.8 J
Top of heap	498	0.33 J	1.0 U	22.2	802	0.63 R	3.3 J	246	0.11 UJ	9.8	238	J 2.5 J	14.3 J
Top of heap	557	0.16 J	1.0 U	30.7	718	0.66 J	4.2 J	208	0.34 UJ	7.8	290	J 0.7 J	6.0 J
Top of heap	606	0.06 J	1.0 U	16.2	634	0.46 J	1.5 J	321	0.24 UJ	5.4	181	J 0.9 J	6.1 J
Top of heap	566	0.07 J	1.0 U	27.9	772	0.31 J	5.0 J	393	0.52 R	10.4	213	J 0.5 J	2.8 J
Top of heap	583	0.04 J	1.0 U	23.9	685	0.48 J	32.7 J	522	0.29 UJ	9.4	552	J 0.6 J	2.9 J
Top of heap	695	0.04 J	1.0 U	25.4	830	0.44 J	1.0 UJ	745	0.61 R	11.0	151	J 0.5 J	2.7 J
S.Toe of heap	980	0.03 J	8.4	16.5	603	0.59 J	11.0 J	870	0.45 UJ	19.9	210	J 10.5 J	85.9 J
S.Toe of heap	758	0.35 J	2.3	29.7	832	0.72 J	11.7 J	1380	0.16 UJ	24.1	416	J 3.6 J	91.8 J
S.Toe of heap	1010	0.12 J	6.5	18.9	698	1.5 J	8.6 J	1400	0.07 R	22.1	561	J 5.5 J	86.1 J
S.Toe of heap	928	J 0.10 J	2.4	11.9 J	668	1.3 J	5.8 J	851	0.01 R	21.6	306	8.7 J	95.1 J
S.Toe of heap	829	J 0.33 J	1.0 U	24.3 J	751	0.01 UJ	13.6 J	1090	0.08 R	15.5	411	15.1 J	173 J
S.Toe of heap	919	---	1.0 U	21.9	723		14.1	1070	6.4	15.9	450		
N. and E. edge	591	J 0.23 J	1.0 U	48.1 J	729	1.1 J	2.7 J	101	0.06 UJ	26.7	149	0.6 J	2.7 J
N. and E. edge	616	J 0.34 J	1.0 U	51.4 J	637	0.97 J	3.5 J	247	0.02 UJ	24.3	187	0.7 J	2.4 J
N. and E. edge	574	J 0.36 J	1.0 U	51.1 J	644	1.6 J	3.2 J	322	0.04 UJ	27.7	175	0.8 J	1.8 J
N. and E. edge	494	J 0.019 UJ	1.0 U	35.5 J	915	0.35 J	1.0 UJ	78	0.14 R	25.9	107	---	0.3 J
N. and E. edge	462	J 0.03 J	1.0 U	36.5 J	1030	0.55 J	39.3 J	80	0.08 UJ	29.8	105	---	0.5 J
N. and E. edge	718	J 0.03 J	1.0 U	41.8 J	602	0.62 J	1.0 UJ	69	0.06 UJ	40.3	127	---	0.3 J
N. and E. edge	619	J 0.05 J	1.0 U	29.4 J	1740 J	0.50 J	1.1 J	169 J	0.11 UJ	26.2	121	0.9 J	3.3 J
N. and E. edge	573	J 0.018 UJ	1.0 U	36.9 J	534 J	0.54 J	5.8 J	134 J	0.04 UJ	24.1	130	---	0.2 J
Mine Dump	311	J 0.54 J	1.0 U	9.0 J	610 J	0.85 J	21.1 J	125 J	0.10 UJ	1.3 U	152	---	0.2 J
Mine Dump	426	J 0.78 J	1.0 U	8.1 J	672 J	0.84 J	13.7 J	104 J	0.04 UJ	1.3 U	150	---	0.2 J
Mine Dump	616	J 0.28 J	1.0 U	46.4 J	592 J	2.0 J	4.8 J	44 J	0.04 UJ	14.3	154	---	0.2 J
Nearby Soil	416	J 0.014 UJ	1.0 U	31.7 J	2000 J	0.65 J	1.4 J	151 J	0.02 UJ	17.7	212	---	0.3 J
Nearby Soil	350	J 0.015 UJ	1.0 U	12.7 J	1670 J	0.14 UJ	1.0 UJ	128 J	0.04 UJ	25.2	53.2	---	0.2
Nearby Soil	612	J 0.15 J	1.0 U	44.9 J	599 J	1.0 J	38.4 J	63 J	0.02 UJ	34.0	133	1.2	4.1
Nearby Soil	524	J 0.05 J	17.3	30.9 J	906 J	0.69 J	6.5 J	87 J	0.01 UJ	20.6	205	1.7	18.2
Nearby Soil	383	J 0.012 UJ	14.3	30.7 J	839 J	1.2 J	1.0 UJ	65 J	0.06 UJ	20.4	193	---	0.3
Background	475	0.02 U	---	16.5	1920	0.12 UJ	2.0 UJ	152	0.62 R	25.3	82.9	---	0.09 J
Background	476	---	---	16.4	1950	3.3		153	0.40	25.3	83.9	---	---
Background	322	0.02 U	---	13.7	1650	0.1 UJ	2.0 UJ	132	0.30 R	28.6	54.8	---	0.11 J

- Notes: 1. All units in milligrams/kilogram of sample (DRY BASIS).
2. Sample no. suffix "D" = duplicate
3. "----" Denotes sample not analyzed.
4. U denotes compound not detected. Value given is the level of quantification.
5. UJ denotes compound not detected. Value given is the estimated level of quantification.
6. J denotes compound quantification is estimated.
7. R denotes data are unusable. Compound may or may not be present.

Table 4.2. EP toxicity results ($\mu\text{g/L}$) and TCLP results ($\mu\text{g/L}$) for leach heap material.

Compound	Sample Numbers					

EP TOXICITY RESULTS (µg/L)						
	01	02	03	04	EXTBLK	EP TOXICITY CRITERIA (mg/L)
-----						-----
Arsenic	3.8 J	18.9	25.8	4.8 J	2.0 UJ	5
Barium	210	36.9	68.0	263	3.0 U	100
Cadmium	5.0	14.6	15.3	4.0 U	4.0 U	1
Chromium	10.0 U	10.0 U	10.0 U	10.0 U	10.0 U	5
Lead	1.0 U	8.9 J	7.5 J	103 J	1.0 U	5
Mercury	0.2 U	0.2 U	0.20	0.2 U	0.2 U	0.2
Selenium	3.0 UJ	3.0 J	15.0 UJ	3.0 UJ	3.0 UJ	1
Silver	8.0 U	8.0 U	8.0 U	8.0 U	8.0 U	5

TCLP RESULTS ($\mu\text{g/L}$)

	14	15	16	17	PB40.35	PB40.36

Arsenic	500 U	500 U	500 U	500 U	500 U	500 U
Barium	571	218	183	180	50 U	50 U
Cadmium	50 U	50 U	50 U	50 U	50 U	50 U
Chromium	100 U	100 U	100 U	100 U	100 U	100 U
Lead	500 U	500 U	544	500 U	500 U	500 U
Mercury	0.06	0.06 U	0.06 U	0.06 U		
Selenium	100 U	100 U	100 U	100 U	100 U	100 U
Silver	50 U	50 U	50 U	50 U	50 U	50 U

Notes: Data qualifier of "U" = the material was analyzed for but not detected. The numerical value shown is the sample quantitation limit.
 Data qualifier of "J" = the associated numerical value is an estimated quantity.
 Data qualifier of "UJ" = the material was analyzed for but not detected. The sample quantitation limit is an estimated quantity.

4.2 EXTENT OF SOIL CONTAMINATION

4.2.1 Location of Sample Sites

Ten soil samples were collected to determine the extent of inorganic contamination of soils adjacent to the leach heap and mine dump. Locations of these samples are shown on Figure 2.3a and laboratory results are included in Table 4.1. Six soil samples were collected from beneath the leachate pond liner. Five composite samples were collected downslope from the pond, near the cattle trough, at the seep, north of the heap, and east of the heap. Additionally, two background composites were collected about 500 feet north and south of the heap.

Ten samples were collected to verify field observations that no organic contamination existed at the site. Sample locations for organic analyses were chosen on the basis of historical information on operations, photographs of the site, and observations of partly filled trenches, stained ground, and an abandoned drum.

4.2.2 Soils Analytical Results

4.2.2.1 Inorganic Results

Soil samples were analyzed for the same inorganic parameters as heap and mine dump materials (Table 2.2). The results are shown in Table 4.1. Sample numbers correlate with sample locations shown in Figure 2.3a.

With the exception of arsenic, the soil samples beneath the pond liner were found to have concentrations of major and minor constituents similar to those found in the heap and mine dump. Concentrations of arsenic were appreciably lower, with a maximum arsenic concentration of 274 mg/kg beneath the pond liner.

Concentrations of several trace elements in soils beneath the liner were also less than those in the heap and dump, especially zinc, copper, and lead. An exception is Sample 6. This sample showed the highest or nearly the highest values for several constituents, including zinc at 800 mg/kg, copper at 504 mg/kg, silver at 30 mg/kg, cadmium at 6.8 mg/kg, and mercury at 1.45 mg/kg. Sample 6 also showed the highest value for total cyanide in soils, at 101 mg/kg. This is second only to 173 mg/kg found at the toe of the heap. Sample 6 indicates the presence of heap-type material, as well as leakage of cyanide beneath the pond liner.

Nearby soil samples were also similar in major and minor elemental composition to the leach heap, with the exception of appreciably lower levels of arsenic and sodium. Among the trace constituents, zinc, copper, lead, mercury, and cyanide were at much lower levels compared to the leach heap.

The two background soil samples differed in most constituents from both the heap material and the nearby soil samples. Except for barium, all trace elements occurred at lower levels in the background samples.

4.2.2.2 Organic Results

Table 4.3 lists the results for organic analyses for soils. Concentrations of most volatile (VOA) and base neutral/acid extractable (BNA) organic parameters included on Table 2.3 were below detection limits. Low levels of acetone and methylene chloride were, however, detected in several of the VOA soil samples at concentrations as high as 10 and 21 $\mu\text{g}/\text{kg}$, respectively. These compounds were also detected in blank samples. The average values for the two parameters are indistinguishable between field samples and blanks at the 90% confidence level (appendix D). Therefore, the site is unlikely to be the source of acetone and methylene chloride in the soil samples.

Low levels of benzyl alcohol, benzoic acid, and bis(2-ethylhexyl)phthalate at concentrations as high as 220, 830, and 4800 $\mu\text{g}/\text{kg}$, respectively, were detected in some soil samples. These compounds represent common laboratory contaminants (Appendix D), however, and are unlikely to represent contamination originating at the site. Most soil samples also contained low concentrations of a variety of organic compounds not included in Table 2.3. The most abundant of these are hydrocarbons at concentrations up to 8400 $\mu\text{g}/\text{kg}$. These "tentatively identified compounds" are listed in the quality assurance reports for organics.

Table 4.3a. Organic analytical results for soils: Volatile organic compounds.

Compound -----	Sample Numbers -----															
	JF002	JF003	JF004	JF005	JF006	JF007	JF008	JF009	JF010	JF011	JF012	JF013	JF014	JF015	JF016	
Chloromethane	10 U	11 U	12 U	12 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	
Bromomethane	10 U	11 U	12 U	12 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	
Vinyl Chloride	10 U	11 U	12 U	12 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	
Chloroethane	10 U	11 U	12 U	12 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	
Methylene Chloride	3 J	5 U	9	9	5 J	5 J	4 J	10	3 J	5 U	4 J	4 J	4 J	4 J	4 J	
Acetone	10 U	11 U	21 J	12 U	10 U	10 U	16 J	10 J	10 U	10 J	10 U	10 U	11 J	10 U	10 U	
Carbon Disulfide	5 U	5 U	6 U	6 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	
1,1-Dichloroethane	5 U	5 U	6 U	6 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	
1,1-Dichloroethene	5 U	5 U	6 U	6 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	
1,1-Dichloroethene (total)	5 U	5 U	6 U	6 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	
Chloroform	5 U	5 U	6 U	6 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	
1,2-Dichloroethane	5 U	5 U	6 U	6 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	
2-Butanone	10 U	11 U	12 U	12 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	
1,1,1-Trichloroethane	5 U	5 U	6 U	6 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	
Carbon Tetrachloride	5 U	5 U	6 U	6 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	
Vinyl Acetate	10 U	11 U	12 U	12 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	
Bromodichloromethane	5 U	5 U	6 U	6 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	
1,2-Dichloropropane	5 U	5 U	6 U	6 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	
cis-1,3-Dichloropropene	5 U	5 U	6 U	6 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	
Trichlororoethene	5 U	5 U	6 U	6 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	
Dibromochloromethane	5 U	5 U	6 U	6 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	
1,1,2-Trichloroethane	5 U	5 U	6 U	6 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	
Benzene	5 U	5 U	6 U	6 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	
Trans-1,3-Dichloropropene	5 U	5 U	6 U	6 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	
Bromoform	5 U	5 U	6 U	6 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	
4-Methyl-2-Pentanone	10 U	11 U	12 U	12 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	
2-Hexanone	10 U	11 U	12 U	12 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	10 U	
Tetrachloroethene	5 U	5 U	6 U	6 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	
1,1,2,2-Tetrachloroethane	5 U	5 U	6 U	6 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	
Toluene	5 U	5 U	6 U	6 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	
Chlorobenzene	5 U	5 U	6 U	6 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	
Ethylbenzene	5 U	5 U	6 U	6 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	
Styrene	5 U	5 U	6 U	6 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	
Total Xylenes	5 U	5 U	6 U	6 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	

Notes: Data qualifier of "U" = the material was analyzed for but not detected. The numerical value shown is the sample quantitation limit.
Data qualifier of "J" = the associated numerical value is an estimated quantity.
Data qualifier of "UJ" = the material was analyzed for but not detected. The sample quantitation limit is an estimated quantity.

Table 4.3b. Soils Analytical Results Semivolatile Organics and Total Organic Carbon

Compound -----	Sample Numbers -----															
	JF002	JF003	JF004	JF005	JF006	JF007	JF008	JF009	JF010	JF011	JF012	JF013	JF014	JF015	JF016	
Phenol	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U	
bis(2-Chloroethyl)Ether	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U	
2-Chlorophenol	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U	
1,3-Dichlorobenzene	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U	
1,4-Dichlorobenzene	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U	
Benzyl Alcohol	220 J	690 U	760 U	770 U	660 U	680 U	670 J	690 U	660 U	220 J	680 U	680 U	220 J	220 J	680 U	
1,2-Dichlorobenzene	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U	
2-Methylphenol	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U	
bis(2-Chloroisopropyl)Ether	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U	
4-Methylphenol	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U	
N-nitroso-Di-n-Propylamie	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U	
Hexaxhloroethane	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U	
Nitrobenzene	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U	
Isophorone	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U	
2-Nitrophenol	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U	
2,4-Dimethylphenol	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U	
Benzoic Acid	830 J	3400 U	200 J	680 J	3200 U	3300 U	3300 U	310 J	3200 U	3200 J	3300 U	3300 U	3300 J	3200 J	3300 U	
bis(2-Chloroethoxy)Methane	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U	
2,4-Dichlorophenol	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U	
1,2,4-Trichlorobenzene	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U	
Naphthalene	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U	
4-Chloroaniline	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U	
Hexachlorobutadiene	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U	
4-Chloro-3-Methylphenol	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U	
2-Methylnaphthalene	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U	
Hexachlorocyclopentadiene	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U	
2,4,6-Trichlorophenol	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U	
2,4,5-Trichlorophenol	3200 U	3400 U	3700 U	3700 U	3200 U	3300 U	3300 U	3300 U	3200 U	3200 U	3300 U	3300 U	3300 U	3200 U	3300 U	
2-Chloronaphthalene	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U	
2-Nitroaniline	3200 U	3400 U	3700 U	3700 U	3200 U	3300 U	3300 U	3300 U	3200 U	3200 U	3300 U	3300 U	3300 U	3200 U	3300 U	
Dimethyl Phthalate	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U	
Acenaphthylene	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U	
2,6-Dinitrotoluene	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U	
3-Nitroaniline	3200 U	3400 U	3700 U	3700 U	3200 U	3300 U	3300 U	3300 U	3200 U	3200 U	3300 U	3300 U	3300 U	3200 U	3300 U	
Acenaphthene	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U	
2,4-Dinitrophenol	3200 U	3400 U	3700 U	3700 U	3200 U	3300 U	3300 U	3300 U	3200 U	3200 U	3300 U	3300 U	3300 U	3200 U	3300 U	
4-Nitrophenol	3200 U	3400 U	3700 U	3700 U	3200 U	3300 U	3300 U	3300 U	3200 U	3200 U	3300 U	3300 U	3300 U	3200 U	3300 U	
Dibenzofuran	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U	
2,4-Dinitrotoluene	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U	
Diethylphthalate	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U	
4-Chlorophenyl-phenylether	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U	

(Continued)

Table 4.3b. (Continued) Soils Analytical Results Semivolatile Organics and Total Organic Carbon

Fluorene	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U
4-Nitroaniline	3200 U	3400 U	3700 U	3700 U	3200 U	3300 U	3300 U	3300 U	3200 U	3200 U	3300 U	3300 U	3300 U	3200 U	3300 U
4,6-Dinitro-2-Methylphenol	3200 U	3400 U	3700 U	3700 U	3200 U	3300 U	3300 U	3300 U	3200 U	3200 U	3300 U	3300 U	3300 U	3200 U	3300 U
N-Nitrosodiphenylamine(1)	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U
4-Bromophenyl-phenylether	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U
Hexachlorobenzene	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U
Pentachlorophenol	3200 U	3400 U	3700 U	3700 U	3200 U	3300 U	3300 U	3300 U	3200 U	3200 U	3300 U	3300 U	3300 U	3200 U	3300 U
Phenanthrene	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U
Anthracene	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U
Di-n-Butylphthalate	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U
Fluoranthene	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U
Pyrene	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U
Butylbenzylphthalate	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U
3,3'-Dichlorobenzidine	1300 U	1400 U	1500 U	1500 U	1300 U	1400 U	1300 U	1400 U	1300 U	1300 U	1400 U	1400 U	1300 U	1300 U	1400 U
Benzo(a)Anthracene	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U
Chrysene	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U
bis(2-Ethylhexyl)Phthalate	1100	690 U	760 U	770 U	310 J	93 J	220 J	690 U	160 J	110 J	680 U	680 U	670 U	110 J	4800
Di-n-Octyl Phthalate	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U
Benzo(b)Fluoranthene	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U
Benzo(k)Fluoranthene	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U
Benzo(a)Pyrene	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U
Indeno(1,2,3-cd)Pyrene	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U
Dibenz(a,h)Anthracene	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U
Benzo(g,h,i)Perylene	670 U	690 U	760 U	770 U	660 U	680 U	670 U	690 U	660 U	670 U	680 U	680 U	670 U	670 U	680 U
Total Organic Carbon	0.98				1.2				0.5		2.2				0.7

Notes: Data qualifier of "U" = the material was analyzed for but not detected. The numerical value shown is the sample quantitation limit.
 Data qualifier of "J" = the associated numerical value is an estimated quantity.
 Data qualifier of "UJ" = the material was analyzed for but not detected. The sample quantitation limit is an estimated quantity.

Toc results are an average of replicate analysis.

4.3 EXTENT OF WATER CONTAMINATION

4.3.1 Location of Sample Sites

Figures 2.3a and 2.5 show the well and surface water sample sites. The sites include 4 onsite monitoring wells, the seep northwest of the heap, mine drainage sampled from the cattle trough, two offsite irrigation wells, and one offsite drinking water well. Details of the monitoring well construction and water sampling methods are discussed in Section 2.8 and 2.9.

4.3.2 Analytical Results

Water samples from three rounds collected in May, June, and July 1989 were analyzed for the inorganic parameters listed in Table 2.2. Data from Rounds 1 and 2 are reported for subsets of the parameter list, whereas data from Round 3 data are reported for the complete list. Before collecting samples for laboratory analysis, water levels and temperature were measured by downhole probes. The remaining field tests, including pH, Eh, specific conductivity, and the presence/absence of free cyanides, sulfides, and oxidants, were conducted immediately after withdrawing water from the wells or surface pools. Most of the field parameters were also measured several times between December 1988 through April 1989. Results of the field measurements are listed in Table 3.3. Laboratory results are listed in Tables 4.4 to 4.8. Water level, temperature, and electrical conductivity are discussed in relation to ground water flow in Section 3.6.1.

Laboratory analyses of the metals listed in Table 2.2 were conducted on filtered and unfiltered samples, to indicate dissolved concentrations and dissolved plus particulate concentrations in water, respectively. The first round of metals samples were not preserved until after receipt by the laboratory. Results on cations from the first round may thus underestimate actual concentrations in the ground water. Metals samples collected for the second round were inadvertently contaminated during preservation. The results of these second round metals analyses are not reported. Metals samples for the third round were preserved by field acidification and are considered to be the most representative of actual ground water conditions. The Round 3 analyses are emphasized for metals in the following Chapters that quantitatively evaluate the significance of the data. Metals results are listed in Tables 4.4 (Round 1) and 4.7 (Round 3).

Anion determinations were made on unfiltered samples.

Cyanide analyses were made of samples collected after first checking for the presence of interfering constituents, including sulfides or oxidants, using the procedure discussed in Chapter 2. Cyanide samples were then preserved with sodium hydroxide. Anion and cyanide results are listed in Tables 4.5 (Round 1), 4.6 (Round 2), and 4.8 (Round 3).

4.3.2.1 Onsite Ground Water

The field parameters (Table 3.3) and the relative proportions of the major dissolved constituents may be used to characterize the general compositional pattern in ground water. Figure 4.3 displays a trilinear Piper diagram (after Piper, 1944) of the proportions in milliequivalents of the major cations (sodium, potassium, calcium, and magnesium) and anions (chloride, sulfate, bicarbonate, and carbonate) for all of the Round 3 analyses (Tables 4.7 and 4.8). The Piper diagram, laboratory analyses, and field parameters indicate that onsite ground water is a neutral to slightly alkaline magnesium-sodium sulfate solution with about 400-900 mg/L total dissolved solids.

The major cations, in order of decreasing concentration, are sodium, magnesium, calcium, and potassium. Samples from Monitoring Well 3, at the southeast corner of the leach heap, showed the highest concentrations of major cations. Samples from Wells 1 and 4 were relatively high in iron and aluminum. The highest values in Round 3 were obtained from Well 1 with 10.7 mg/L of iron and 9.3 mg/L of aluminum. Other dissolved constituents measured at relatively high concentrations in Round 3 samples include manganese at 270 $\mu\text{g/L}$ and copper at 48 $\mu\text{g/L}$ in Well 1, and antimony at 44 $\mu\text{g/L}$ in Well 2.

The major anions in onsite ground water, in order of decreasing abundance, are sulfate, bicarbonate, nitrate, and chloride. The Piper diagram (Figure 4.3) indicates a somewhat higher proportion of sulfate in onsite ground water relative to offsite water. However, no compositional trend comparable to that for cations is apparent for the anions shown in the diagram.

In Round 3 the highest nitrate concentration occurred in samples from Well 3, at 17 mg/L. Much greater concentrations of nitrate, the highest at 120 mg/L, were found in Round 2 samples from Wells 1, 2, and 3. Round 2 values are qualified as estimates, however, and neither the high concentrations nor the distribution pattern of nitrate from Round 2 were verified in Round 3. Thus the nitrate values from Round 3 are considered to be the most representative.

For all three rounds, total cyanide concentrations are

consistently highest in Well 3, with values ranging from 30 $\mu\text{g/L}$ in Round 2 to 280 $\mu\text{g/L}$ in Round 3. Corresponding weak acid dissociable cyanide concentrations for Well 3 range from 3.1 to 92 $\mu\text{g/L}$.

Two divergent distribution patterns of major constituents and contaminants are apparent from the Round 3 data. An increase in concentration of several parameters occurs in the downgradient direction as ground water passes beneath the heap. A representative example of this pattern is the distribution of cyanide, nitrate, and electrical conductivity shown in Figure 4.4. All are highest to the southeast in well 3 which is the downgradient direction. This pattern is consistent with the compositional trend for major cations showing increasing proportion of sodium and potassium in the downgradient direction from wells 1 and 2 to wells 3 and 4 (Figure 4.3). Other parameters showing the same trend include fluoride and nitrite.

On the other hand, other parameters show higher concentrations in wells 1 and 2, which lie respectively upgradient and marginal to the downgradient direction from the heap. Included in this set of parameters are arsenic, antimony, barium, chloride, chromium, copper, iron, lead, manganese, nickel, silver, and zinc. The concentrations of arsenic are displayed in figure 4.4.

4.3.2.2 Onsite Surface Water

In contrast to the onsite ground water, both the seep and the mine drainage are slightly alkaline magnesium sulfate solutions (Tables 4.7, 4.8). As shown by the Piper diagram (Figure 4.3), sodium and potassium occur in much lower proportions in the surface water than in the ground water. With the exception of arsenic, elevated levels of most constituents do not occur in the surface water. Although cyanide was detected at 1.2 $\mu\text{g/L}$ in a Round 2 sample of the mine drainage, it was below detection limits in the other two rounds.

Arsenic, on the other hand, is higher in the mine drainage than in any other water at this site. Dissolved arsenic concentration in the mine drainage was 91 $\mu\text{g/L}$ in Round 3.

4.3.2.3 Offsite Ground and Surface Water

Samples from the three wells within three miles downgradient of the site indicate that ground water from the main portion of the Horse Springs Coulee-Aeneas Lake aquifer is a calcium-magnesium bicarbonate solution (Figure 4.3) varying from neutral

to slightly alkaline. The offsite ground water samples were lower in the proportion of sodium, potassium, and sulfate in comparison with onsite water. The offsite water also had lower concentrations of all other major constituents (Table 4.7, 4.8) and did not contain elevated levels of contaminants. Overall, water drawn at the offsite wells appears to be of good quality.

The quality of offsite surface water was not investigated. The closest discharge of water from the area of the mine site to an offsite surface water body is likely no closer than Aeneas Lake, five miles to the southeast (Russell and Eddy, 1971). The flow path would be from onsite, through the Horse Springs Coulee aquifer, to Aeneas Lake. The three water supply wells examined during the Remedial Investigation lie between the mine site and the lake. The water quality in these wells is expected to be more indicative of ground water directly upgradient of any discharge to surface water (such as Aeneas Lake) than onsite ground water. As noted above, these wells have generally good water quality. Therefore, onsite contaminants are unlikely to extend at present to either offsite ground water or surface water.

RI CHAPTER 4
PAGE 17

Table 4.4. Water Analytical Results. Round 1 Metals.

Location	WFOC #	Al	Sb	As	Ba	Be	Cd	Ca	Cr	Co	Cu	Fe	Pb
Monitoring Well 1F	890515041	6000	2.2 J	4.9 R	72.2 J	1.0 UJ	1.0 U	43500	9.5	3.9	27.0 J	6530	16.4 R
Monitoring Well 1U		8670	3.0 R	6.9 R	78.9 J	1.0 UJ	1.0 U	44900	15.3	6.8	41.7 J	10200	20.5 R
Monitoring Well 2F	890515042	188	77.7 J	17.6 R	32.2 J	1.0 UJ	1.0 U	51000	2.8	2.0 U	23.1 J	540	20.2 R
Monitoring Well 2U		281	59.0 J	17.9 R	33.9 J	1.0 UJ	1.0 U	51100	3.6	2.5 U	20.0 J	587	17.1 R
Monitoring Well 3F	890515043	36.3	1.7 J	2.0 R	42.6 J	1.0 UJ	1.0 U	89300	2.3	7.5	14.2 J	102	13.5 R
Monitoring Well 3U		37.5	175 R	5.0 R	30.1 J	1.0 UJ	1.0 U	87200	2.0 U	6.1	11.9 J	92.5	9.3 R
Monitoring Well 4F	890515044	2720	5.4 J	13.1 R	59.9 J	1.0 UJ	1.0 U	22100	6.8	2.5 U	13.4 J	2050	12.7 R
Monitoring Well 4U		1620	6.2 J	17.0 R	58.0 J	1.0 UJ	1.0 U	22300	3.1	2.5 U	16.2 J	1310	18.0 R
Stock TankF	890515045	22.6	10.5 J	83.5 R	13.4 J	1.0 UJ	1.0 U	44800	3.0	2.5 U	14.9 J	52.7	11.5 R
Stock TankU		24.7	32.0 J	87.3 R	8.8 J	1.0 UJ	1.0 U	43800	14.5	2.5 U	12.5 J	110	8.6 R
Mine SeepF	890515046	96.3	1.0 UJ	2.3 R	14.6 J	1.0 UJ	1.0 U	64200	2.6	1.0 U	16.6 J	343	11.9 R
Mine SeepU		277	0.7 UJ	4.5 R	17.0 J	1.0 UJ	1.0 U	66000	3.3	2.5 U	14.4 J	495	11.2 R
Irrigation Well 1F	890515047	10.8	0.1 UJ	2.0 R	22.4 J	1.0 UJ	1.0 U	50500	4.1	2.5 U	13.8 J	45.0	20.6 R
Irrigation Well 1U		15.8	0.8 UJ	2.0 R	24.0 J	1.0 UJ	1.0 U	50200	6.0	2.5 U	14.9 J	169	8.3 R
Irrigation Well 2F	890515048	10.0	0.1 UJ	6.1 R	30.0 J	1.0 UJ	1.0 U	67400	5.0	2.5 U	13.9 J	54.3	10.7 R
Irrigation Well 2U		11.7	108 J	9.2 R	29.0 J	1.0 UJ	1.0 U	65300	2.8	2.5 U	14.7 J	146	12.7 R
RinsateF	890515049	56.3	0.3 UJ	2.0 R	4.8 J	1.0 UJ	1.0 U	391	6.1	2.5 U	24.8 UJ	107	J 12.2 R
RinsateU		46.7	0.1 UJ	2.0 R	3.4 J	1.0 UJ	1.0 U	354	4.4	2.5 U	23.8 UJ	133	J 14.2 R
RinsateFD		76.8		2.0	5.1	0.6	0.9	397	5.5	0.1	64.4	176	19.1
RinsateF	890515050	32.4	0.1 UJ	3.5 R	10.0 J	1.0 UJ	1.0 U	628	5.3	2.5 U	23.4 UJ	140	J 20.6 R
RinsateU		26.8	0.1 UJ	2.4 R	3.4 J	1.0 UJ	1.0 U	351	8.6	2.5 U	31.0 UJ	102	J 13.0 R
RinsateF	890515051	35.3	0.1 UJ	2.0 R	2.4 J	1.0 UJ	1.0 U	592	16.6	2.5 U	26.4 UJ	157	J 9.4 R
RinsateU		57.0	0.1 UJ	2.6 R	3.4 J	1.0 UJ	1.0 U	954	32.2	2.5 U	51.9 UJ	812	J 89.5 R
Deionized WaterF	890515052	24.7	17.5 UJ	2.9 R	2.0 J	1.4 UJ	1.0 U	105	3.6 U	2.5 U	30.1 UJ	72.9 J	21.0 R
Deionized WaterU		21.1	0.1 UJ	2.0 R	6.0 J	1.0 UJ	1.0 U	173	2.0 U	2.5 U	29.9 UJ	60.0 J	27.9 R

Note: 1. All concentrations are in $\mu\text{g/L}$ (ppb).

2. Suffix "F" = filtered, "U" = unfiltered, "D" = duplicate

3. U denotes compound not detected. Numerical value given is the level of quantification.

4. UJ denotes compound not detected. Numerical value given is the estimated level of quantification.

5. J denotes that the numerical value given is an estimated quantity.

6. R denotes data are unusable. Compound may or may not be present.

RI CHAPTER 4
PAGE 18

Table 4.4. (Continued)

Location		Mg	Mn	Hg	Mo	Ni	K	Se	Si	Ag	Sodium	Tl	Sn	V	Zn
Monitoring Well 1F	41500	129	0.20 UJ	10	13.3	3030 J	2.3 J	100	U	2.0 UJ	75100 J	0.1 UJ	9	U 15.6	80.3
Monitoring Well 1U	43600	200	0.20 UJ	8	18.2	3630 J	4.1 J	100	U	2.0 UJ	84700 J	0.1 UJ	9	U 23.1	115
Monitoring Well 2F	90900	86.7	0.20 UJ	10	8.7	3450 J	0.4 UJ	100	U	3.4 UJ	52400 J	0.1 UJ	9	U 2.5	U 72.9
Monitoring Well 2U	93700	113	0.20 UJ	12	8.5	3450 J	0.3 UJ	100	U	2.0 UJ	52600 J	0.1 UJ	9	U 2.5	U 67.9
Monitoring Well 3F	80200	49.3	0.20 UJ	12	7.1	4400 J	6.3 J	100	U	2.0 UJ	103000 J	0.1 UJ	11	2.5	U 32.1
Monitoring Well 3U	79800	47.8	0.20 UJ	15	7.2	5490 J	6.2 J	100	U	2.0 UJ	137000 J	0.1 UJ	9	U 2.5	U 24.8
Monitoring Well 4F	14100	40.1	0.20 UJ	18	7.4	2750 J	1.2 UJ	100	U	2.0 UJ	60700 J	0.1 UJ	9	U 4.6	26.9
Monitoring Well 4U	14400	31.2	0.20 UJ	17	6.1	3090 J	1.1 UJ	100	U	2.0 UJ	67200 J	0.1 UJ	9	U 3.5	37.6
Stock TankF	73100	2.3	0.20 UJ	6	10.7	1160 J	1.2 UJ	100	U	2.0 UJ	19800 J	0.7 R	9	U 2.5	U 30.1
Stock TankU	71400	3.8	0.20 UJ	6	14.0	1230 J	1.3 J	100	U	2.0 UJ	21000 J	0.3 R	9	U 2.5	U 22.6
Mine SeepF	74900	5.3	0.20 UJ	5	U 4.0	1520 J	5.8 J	100	U	2.0 UJ	21800 J	0.1 R	4	2.5	U 22.9
Mine SeepU	75300	7.5	0.20 UJ	5	U 5.0	1630 J	4.9 J	100	U	2.0 UJ	21000 J	0.3 R	9	U 2.5	U 17.3
Irrigation Well 1F	26100	2.0	0.20 UJ	5	U 2.3	3250 J	1.7 J	100	U	2.0 UJ	20100 J	0.4 R	9	U 2.5	U 31.8
Irrigation Well 1U	26000	2.0	0.20 UJ	5	U 4.2	3270 J	1.9 J	100	U	2.0 UJ	20700 J	0.2 R	9	U 2.5	U 19.6
Irrigation Well 2F	19700	2.0	0.20 UJ	5	U 3.5	2500 J	1.7 J	100	U	2.3 UJ	14500 J	0.5 R	9	U 2.5	U 31.2
Irrigation Well 2U	19400	2.0	0.20 UJ	5	U 3.7	3280 J	1.9 J	100	U	2.0 UJ	18600 J	0.6 R	0	2.5	U 35.0
RinsateF	153	2.9	0.36 UJ	5	U 12.9 J	398 J	0.1 UJ	100	U	2.0 UJ	699	0.2 UJ	0	2.5	U 65.4
RinsateU	146	2.9	0.20 UJ	5	U 14.9 J	351 J	0.1 UJ	100	U	2.0 UJ	604	0.2 UJ	9	U 2.5	U 37.6
RinsateFD	156	4.7		1	U 77.9	453		100	U	0.0	794		1	0.12	51.0
RinsateF	464	3.2	0.30 UJ	5	U 98.8 J	139 J	0.1 UJ	100	U	0.2 UJ	660	0.6 UJ	9	U 2.5	U 42.3
RinsateU	202	3.1	0.20 UJ	5	U 27.9 J	201 J	0.1 UJ	100	U	2.0 UJ	461	0.4 UJ	2	2.5	U 43.3
RinsateF	490	3.9	0.20 UJ	5	U 14.3 J	530 J	0.1 UJ	100	U	2.0 UJ	785	0.3 UJ	0	2.5	U 59.1
RinsateU	1100	8.7	0.20 UJ	5	U 42.4 J	527 J	0.1 UJ	100	U	2.9 UJ	886	0.3 UJ	9	U 2.5	U 170
Deionized WaterF	100 U	2.2	0.20 UJ	5	U 30.3 J	100 U	0.1 UJ	100	U	2.0 UJ	100 U	0.6 UJ	2	2.5	U 51.9
Deionized WaterU	100 U	2.3	0.20 UJ	5	U 25.4 J	100 U	0.1 UJ	100	U	2.0 UJ	110	0.6 UJ	9	U 2.5	U 45.5

Note: 1. All concentrations are in $\mu\text{g/L}$ (ppb).

2. Suffix "F" = filtered, "U" = unfiltered, "D" = duplicate

3. U denotes compound not detected. Numerical value given is the level of quantification.

4. UJ denotes compound not detected. Numerical value given is the estimated level of quantification.

5. J denotes that the numerical value given is an estimated quantity.

6. R denotes data are unusable. Compound may or may not be present.

Table 4.5. Water Analytical Results. Round 1 Anions and Cyanide.

Location	WFOC #	Cl	F	SO4	CN(Total)	CN(WAS)
Monitoring Well 1	890515041	7.5	0.47 U	220	1.9 R	-----
Monitoring Well 2	890515042	4.4	0.77 U	541	3.2 R	-----
Monitoring Well 3	890515043	5.4	0.82 U	770	40.2 R	-----
Monitoring Well 4	890515044	3.0	0.31 U	100	7.0 R	-----
Monitoring Well 4D		3.0	0.29	126		
Stock Tank	890515045	1.9	0.34 U	252	0.2 R	-----
Mine Seep	890515046	1.8	0.45 U	274	0.2 R	-----
Irrigation Well 1	890515047	1.4	0.36 U	110	0.4 R	-----
Irrigation Well 2	890515048	1.4	0.12 U	128	0.2 R	-----
Rinsate	890515049	0.26	0.03 U	1.7	0.2 R	-----
Rinsate	890515050	0.17	0.02 U	4.3	0.2 R	-----
Rinsate	890515051	0.51	0.02 U	3.6	0.2 R	-----
Deionized Water	890515052	0.10	0.11 U	1.3	0.2 R	-----

- Notes: 1. F, Cl and SO4 values are in milligrams/kilogram of sample.
2. Total cyanides reported as micrograms/liter (ppb)-distillation method
3. "-----" Denotes sample not analyzed.
4. U denotes compound not detected. Numerical value given is the level of quantification.
5. R denotes data are unusable. Compound may or may not be present.

Table 4.6. Water Analytical Results. Round 2 Anions and Cyanide.

Location	WFOC #	Cl	F	NO3	SO4	CN(Total)	CN(was)
Monitoring Well 1	890607055	6.4 J	0.26	121 J	156 J	0.80	---
Rinsate	890607056	0.15 J	0.03 U	0.18 J	0.19 J	0.40	---
Monitoring Well 2	890607057	3.0 J	0.36	98.5 J	315 J	2.6	---
Rinsate	890607058	0.14 J	0.03 U	0.16 UJ	0.19 J	0.80	---
Monitoring Well 3	890507059	3.1 J	0.54	42.5 J	482 J	30.6	3.2
Monitoring Well 3D		---	---	---	---	---	3.1
Rinsate	890607060	0.12 J	0.03 U	0.16 UJ	0.18 J	0.80	---
Monitoring Well 4	890607061	1.1 J	0.31	8.2 J	87.4 J	0.01 U	---
Stock Tank	890607062	1.5 J	0.16	0.25 J	168 J	1.2	---
Stock Tank D		1.5 J	0.15	0.25 J	169 J	---	---
Mine Seep	890607063	1.1 J	0.22	0.23 J	173 J	0.01 U	---
Monitoring Well 3	890607064	2.4 J	0.65	34.1 J	358 J	31.4	3.0
Monitoring Well 3D		---	---	---	---	32.8	---
Irrigation Well 2	890907066	0.88 J	0.07	1.1 J	35.9 J	0.80	---
Residence	890607067	0.88 J	0.12	15.5 J	40.0 J	0.40	---
Transport Blank		0.07 UJ	0.03 U	0.16 UJ	0.13 UJ	0.01 U	---

- Note: 1. F, Cl, NO3 and SO4 concentrations are in mg/L (ppm)
2. Suffix "F" = filtered, "U" = unfiltered, "D" = duplicate
3. Cyanide concentrations are in µg/L (ppb)-distillation method
4. "----" Denotes sample not analyzed.
5. Nitrate reported as NO3.
6. U denotes compound not detected. Numerical value given is the level of quantification.
7. UJ denotes compound not detected. Numerical value given is the estimated level of quantification.
8. J denotes that the numerical value given is an estimated quantity.

RI CHAPTER 4
PAGE 21

Table 4.7. Water Analytical Results. Round 3 Metals.

Location	WFOC #	Al	Sb	As	Ba	Be	Cd	Ca	Cr	Co	Cu	Fe	Pb
Monitoring Well 1F	890727069	9250 J	0.1 UJ	9.1 J	91 J	1.0 UJ	1	43500 J	17.1	7.5 UJ	47.6	10700	12.2
Monitoring Well 1U		12000 J	1.4 J	10.9 J	136 J	1.5 J	2	62600 J	31.6	9.4 UJ	56.7	16900	23.2
RinsateF	890727070	77 J	0.5 UJ	0.1 UJ	2 UJ	1.0 UJ	1 U	170 J	2.2 U	2.5 UJ	5.0 U	80	5.0 U
RinsateU		18 J	0.1 UJ	0.1 UJ	2 J	1.0 J	1 U	235 J	2.0 U	2.5 UJ	5.0 U	58	5.0 UJ
Monitoring Well 2F	890727071	213 J	44.0 J	15.3 J	49 J	1.0 UJ	2	31900 J	13.1	3.0 UJ	5.0 U	511	7.7
Monitoring Well 2U		168 J	40.4 J	14.3 J	32 J	1.0 UJ	1	40600 J	2.5	2.5 UJ	11.1	377	6.5
RinsateF	890727072	20 J	0.1 UJ	0.2 UJ	2 J	1.0 UJ	1 U	137 J	2.0 U	2.5 UJ	5.0 U	35	5.0 U
RinsateU		44 J	0.1 UJ	0.2 UJ	2 J	1.0 UJ	1	450 J	2.0 U	2.5 UJ	5.0 U	38	5.0 U
Monitoring Well 3F	890727073	101 J	15.7 J	3.7 J	22 J	1.0 UJ	1 J	67600 J	3.3 U	9.3 UJ	5.4	160	5.0 U
Monitoring Well 3U		106 J	2.3 J	3.5 J	22 J	1.0 UJ	1 J	57800 J	2.0 U	2.5 UJ	10.3	164	5.0 U
Monitoring Well 3DF	890727074	59 J	15.5 J	3.3 J	22 J	1.0 UJ	1 UJ	43900 J	2.0 U	3.9 UJ	5.4	160	5.0 U
Monitoring Well 3DU		93 J	21.2 J	3.9 J	17 J	1.0 UJ	2 J	43100 J	2.0 U	5.3 UJ	6.9	245	5.0 U
RinsateF	890727075	14 J	3.7 J	0.1 UJ	2 UJ	1.0 UJ	1 UJ	137 UJ	2.0 U	2.5 UJ	5.0 U	35	5.0 UJ
RinsateU		15 J	20.7 J	0.1 UJ	2 J	1.0 UJ	1 UJ	100 UJ	3.5	2.5 UJ	5.0 U	32	5.0 UJ
Monitoring Well 4F	890727076	1160 J	2.7 R	13.9 J	51 J	1.0 UJ	1 UJ	22400 J	2.5	2.5 UJ	5.0 U	1390	5.0 U
Monitoring Well 4U		164 J	6.1 R	14.2 J	52 J	1.1 J	3 J	17400 J	9.5	2.5 UJ	5.0 U	206	5.0 U
Stock TankF	890727077	26 J	4.9 R	91.0 J	9 J	1.0 UJ	1 U	45200	2.0 U	2.5 U	5.0 U	52 U	5.0 U
Stock TankU		55 J	4.5 R	95.0 J	10 J	1.0 UJ	1 U	25100	2.0 U	2.0 U	5.0 U	62 U	5.0 U
Mine SeepF	890727078	575 J	0.7 R	4.8 J	16 J	1.0 UJ	1 U	56500	4.3 J	2.5 U	7.6	1040	5.0 U
Mine SeepU		3000 J	1.0 R	8.6 J	38 J	0.9 UJ	1	68400	3.4 J	2.5 U	19.0	4880	7.1
Irrigation Well 1F	890727079	12 J	0.1 UJ	1.6 J	25 J	1.0 UJ	1 U	42000	4.7 U	2.5 U	9.5	43 U	5.0 U
Irrigation Well 1U		10 J	0.1 UJ	2.0 J	27 J	1.0 UJ	1 U	48800	2.0 U	2.5 U	5.0 U	77 U	5.0 UJ
Irrigation Well 2F	890727080	10 U	0.1 UJ	1.2 UJ	24 J	1.0 UJ	1 U	61600	2.0 U	2.5 U	5.0 U	29 U	5.0 U
Irrigation Well 2U		10 U	0.1 UJ	1.3 J	28 J	1.0 UJ	1 U	57900	2.0 U	2.5 U	5.0 U	32 U	5.0 UJ
ResidenceF	890727081	20 U	0.1 UJ	1.0 UJ	19 J	1.0 UJ	1 U	71400	2.5 U	2.5 U	5.0 U	57 U	5.0 U
ResidenceU		17 U	0.1 UJ	1.0 UJ	18 J	1.0 UJ	1 U	47200	2.0 U	2.5 U	5.0 U	26 U	5.0 UJ
Deionized WaterF	890727082	13 U	0.1 UJ	0.3 UJ	4 J	1.0 UJ	1 U	178 U	2.2 U	2.5 U	5.0 U	40 U	5.0 UJ
Deionized WaterU		10 U	0.1 UJ	0.1 UJ	1 J	1.0 UJ	1 U	108 U	2.0 U	2.5 U	5.0 U	25 U	5.0 UJ
Transport BlankU	ALRC1	21 U	0.1 UJ	0.3 UJ	1 UJ	1.1 J	1 U	258 U	2.0 U	2.5 U	5.0 U	36 U	5.0 U
Transport BlankU	ALRC2	10 U	0.1 UJ	0.1 UJ	1 UJ	1.0 UJ	1 U	125 U	2.0 U	2.5 U	5.0 U	38 U	5.0 UJ

- Note: 1. All concentrations are in $\mu\text{g/L}$.
2. Sample location suffix "F" = filtered, "U" = unfiltered, "D" = duplicate
3. "----" denotes sample not analyzed.
4. U denotes compound not detected. Numerical value given is the level of quantification.
5. UJ denotes compound not detected. Numerical value given is the estimated level of quantification.
6. J denotes that the numerical value given is an estimated quantity.
7. R denotes data are unusable. Compound may or may not be present.

RI CHAPTER 4
PAGE 22

Table 4.7. (Continued)

Location	Mg	Mn	Hg	Mo	Ni	K	Se	Ag	Si	Na	Tl	Sn	V	Zn
Monitoring Well 1F	63200 J	271	0.2 U	20.0	22.4	3620	J 3.7 J	7.5	4840	69400 J	1.1 UJ	31.5	28.9	117
Monitoring Well 1U	60900 J	421	0.2 U	10.0 U	38.4	4050	J 3.9 J	5.0 U	---	53800 J	0.1 UJ	20.0 U	40.7	129 J
RinsateF	100 U	2	0.2 U	10.0 U	2.0	100	U 0.2 UJ	5.0 U	292	114 U	1.6 UJ	20.0 U	6.8	13.4
RinsateU	100 U	2	0.2 U	10.0 U	2.1	100	U 0.2 UJ	5.0 U	---	192 U	0.4 UJ	20.0 U	5.2	4.5 J
Monitoring Well 2F	97500 J	93	0.2 U	19.0	10.1	2840	J 0.5 UJ	5.0 U	5100	39900 J	1.1 UJ	26.0	2.5 U	33.7
Monitoring Well 2U	94700 J	77	0.2 U	18.5	9.3	2670	J 0.7 UJ	5.0 U	---	38100 J	0.1 UJ	31.5	2.5 U	30.3 J
RinsateF	100 U	2	0.2 U	10.0 U	2.0	100	U 0.1 UJ	5.0 U	47	173 U	0.5 UJ	20.0 U	2.5 U	7.1
RinsateU	412	2	0.2 U	10.0 U	2.5	100	U 0.1 UJ	5.0 U	---	277 U	1.0 UJ	20.0 U	2.5 U	6.1 J
Monitoring Well 3F	67600 J	123	0.2 U	25.5	6.1	4540	J 2.3 J	5.0 U	5430	117000 J	1.2 R	67.0	5.5 U	10.1
Monitoring Well 3U	69000 J	122	0.2 U	20.5	5.5	4650	J 2.3 J	5.0 U	---	119000 J	1.5 R	20.0 U	2.5 U	3.9 J
Monitoring Well 3DF	67700 J	123	0.2 U	24.0	5.6	4630	J 1.8 J	5.0 U	5410	119000 J	2.5 R	37.5	2.5 U	8.0
Monitoring Well 3DU	74500 J	134	0.2 U	25.0	5.6	4800	J 2.5 J	5.0 U	---	121000 J	0.7 R	20.0 U	2.5 U	5.6 J
RinsateF	100 U	2 U	0.2 U	10.0 U	2.6	100	U 0.1 UJ	5.0 U	42	149 U	0.5 R	20.0 U	2.5 U	6.5
RinsateU	100 U	2 U	0.2 U	10.0 U	3.3	100	U 0.1 UJ	5.0 U	---	112 U	1.0 R	20.0 U	2.9	7.3 J
Monitoring Well 4F	18200 J	62	0.2 U	19.5	4.7	2500	J 0.2 UJ	5.0 U	24	45400 J	0.1 R	23.5	3.5	7.5
Monitoring Well 4U	16100 J	46	0.2 U	12.0	9.1	1620	J 0.4 UJ	5.0 U	---	31700 J	0.7 R	20.0 U	2.8	5.3 J
Stock TankF	82200	2	0.2 U	12.0	10.7 U	984	J 1.6 R	5.0 U	5910	15900 J	1.3 UJ	20.0 U	2.5 U	9.4
Stock TankU	76200	2	0.2 U	10.0 U	13.4	914	J 1.7 R	5.0 U	---	15400 J	0.1 UJ	20.0 U	2.5 U	5.9 J
Mine SeepF	80100	19	0.2 U	10.0 U	7.9 U	1630	J 4.1 R	5.0 U	4170	17400 J	0.1 UJ	20.0 U	2.5 U	24.3
Mine SeepU	78100	41	0.2 U	10.0 U	16.2	2390	J 10.1 R	5.0 U	---	16400 J	0.1 UJ	20.0 U	2.6	56.5 J
Irrigation Well 1F	30700	2 U	0.2 U	12.0	3.2 U	2530	J 1.5 R	5.0 U	16	16000 J	0.6 UJ	20.0 U	2.5 U	4.8
Irrigation Well 1U	32300	2 U	0.2 U	10.0 U	2.0 U	2890	J 1.2 R	5.0 U	---	18500 J	0.1 UJ	20.0 U	2.0 U	2.0 UJ
Irrigation Well 2F	21100	2 U	0.2 U	10.0 U	2.3 U	2460	J 2.5 J	5.0 U	9090	12900 J	0.1 UJ	20.0 U	2.5 U	23.1 J
Irrigation Well 2U	20300	2 U	0.2 U	10.0 U	2.1 U	2430	J 2.3 J	5.0 U	---	12500 J	0.8 UJ	20.0 U	2.5 U	13.5 J
ResidenceF	27300	2 U	0.2 U	10.0 U	5.3 U	2580	J 2.6 J	5.0 U	87	18000 J	0.1 UJ	54.0	2.5 U	4.6 J
ResidenceU	22600	2 U	0.2 U	10.0 U	2.0 U	2590	J 2.4 J	5.0 U	---	18000 J	0.1 UJ	22.5	2.5 U	2.0 UJ
Deionized WaterF	100 U	2 U	0.2 U	10.0 U	4.3 U	100	UJ 0.1 UJ	5.0 U	3	141 UJ	0.1 UJ	20.0 U	2.5 U	3.7 J
Deionized WaterU	100 U	2 U	0.2 U	10.0 U	3.4 U	100	UJ 0.1 UJ	5.0 U	---	100 UJ	0.7 UJ	20.0 U	2.5 U	2.0 UJ
Transport BlankU	100 U	2 U	0.2	10.0 U	3.6 U	100	UJ 0.2 UJ	5.0 U	26	108 UJ	0.2 UJ	20.0 U	2.5 U	8.2 J
Transport BlankU	100 U	2 U	0.2	10.0 U	3.6 U	100	J 0.1 UJ	5.0 U	0	108 UJ	0.1 UJ	20.0 U	2.5 U	2.0 UJ

- Note: 1. All concentrations are in µg/L.
2. Sample location suffix "F" = filtered, "U" = unfiltered, "D" = duplicate
3. "----" denotes sample not analyzed.
4. U denotes compound not detected. Numerical value given is the level of quantification.
5. UJ denotes compound not detected. Numerical value given is the estimated level of quantification.
6. J denotes that the numerical value given is an estimated quantity.
7. R denotes data are unusable. Compound may or may not be present.

RI CHAPTER 4
PAGE 23

Table 4.8. Water Analytical Results. Round 3 Anions and Cyanide.

Location	WFOC #	Alkalinity	Ammonium	Chloride	Fluoride	Nitrite	Nitrate	Sulfate	CN (Tot.)	CN (WAS)
Monitoring Well 1	890727069	364	0.02 U	6.9	0.41	0.01 U	5.5	209	0.6	---
Rinsate	890727070	0.11	0.02 U	0.06	0.01 U	0.01 U	0.01	0.49	0.1 U	---
Monitoring Well 2	890727071	230	0.02 U	3.6	0.50	0.01 U	3.0	434	1.6	---
Rinsate	890727072	0.07	0.02 U	0.06	0.01	0.01 U	0.01 U	1.9	0.1 U	---
Monitoring Well 3	890727073	220	0.02 U	3.7	0.69	1.3	15.9	536	281	43.0 J
Monitoring Well 3D	890727074	226	0.02 U	3.8	0.73	1.4	17.1	563	283	91.6 J
Rinsate	890727075	0.05	0.02 U	0.07	0.01 U	0.01 U	0.06	1.8	0.5	---
Monitoring Well 4	890727076	232	0.02 U	3.6	0.11	0.01 U	2.0	183	9.4	---
Stock Tank	890727077	241	0.02 U	2.4	0.25	0.01 U	0.12 U	407	0.1 U	---
Mine Seep	890727078	347	0.02 U	2.6	0.34	0.01 U	0.01 U	251	0.1 U	---
Irrigation Well 1	890727079	199	0.02 U	2.2	0.20	0.01 U	2.8	122	0.1 U	---
Irrigation Well 2	890727080	215	0.02 U	1.8	0.16	0.01 U	2.8	82.4	0.1 U	---
Residence	890727081	218	0.02 U	1.9	0.13	0.01 U	3.3	117	0.1 U	---
Deionized Water	890727082	0.06	0.02 U	0.04 U	0.01 U	0.01 U	0.01 U	0.40	0.1 U	---
Transport Blank	890727083	0.08	0.02 U	0.04 U	0.01 U	0.01 U	0.01 U	0.03 U	0.12	---
Transport Blank	890727084	0.05	0.02 U	0.04 U	0.01 U	0.01 U	0.11	0.03 U	0.12	---

- Notes: 1. All anion concentrations are in mg/L or ppm, except cyanides which are in µg/L.
2. Alkalinities are in units of mg/L calcium carbonate.
3. "----" Denotes sample not analyzed.
4. Nitrate and nitrite are reported as NO₃ and NO₂ respectively.
5. U denotes compound not detected. Numerical value given is the level of quantification.
6. R denotes data are unusable. Compound may or may not be present.

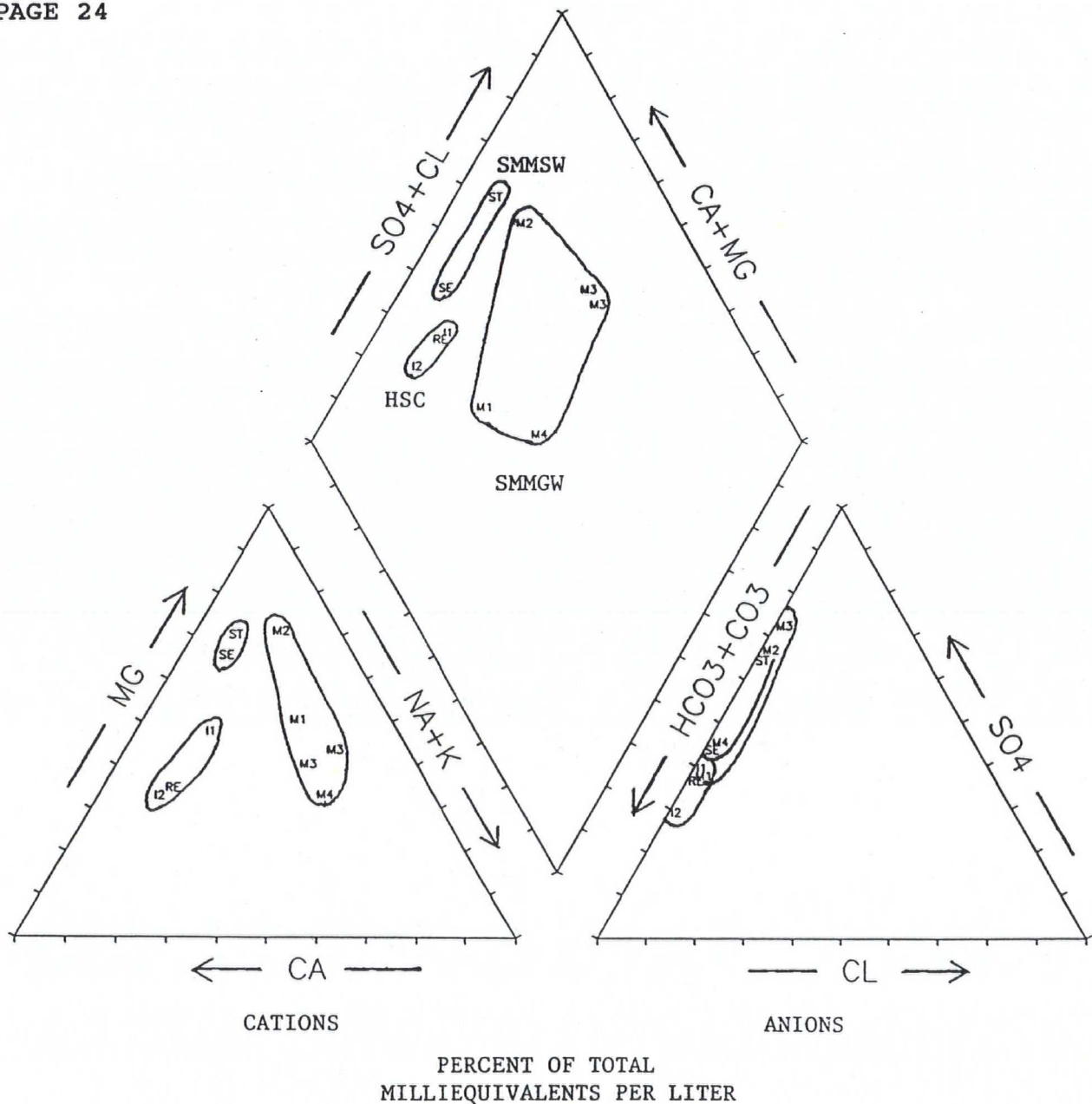


Figure 4.3. Piper diagram of percentage milliequivalents of major cations (left triangle), anions (right triangle), and combined ions (diamond). Values are grouped according to type of sample locality: SMM GW-ground water in shallow aquifer at Silver Mountain Mine where M1, M2, M3, and M4 represent monitoring wells; SMM SW-onsite surface water where ST is stock pond and SE is seep; and HSC-offsite ground water in Horse Springs Coulee where I1 and I2 are irrigation wells and RE is a residential water supply well.

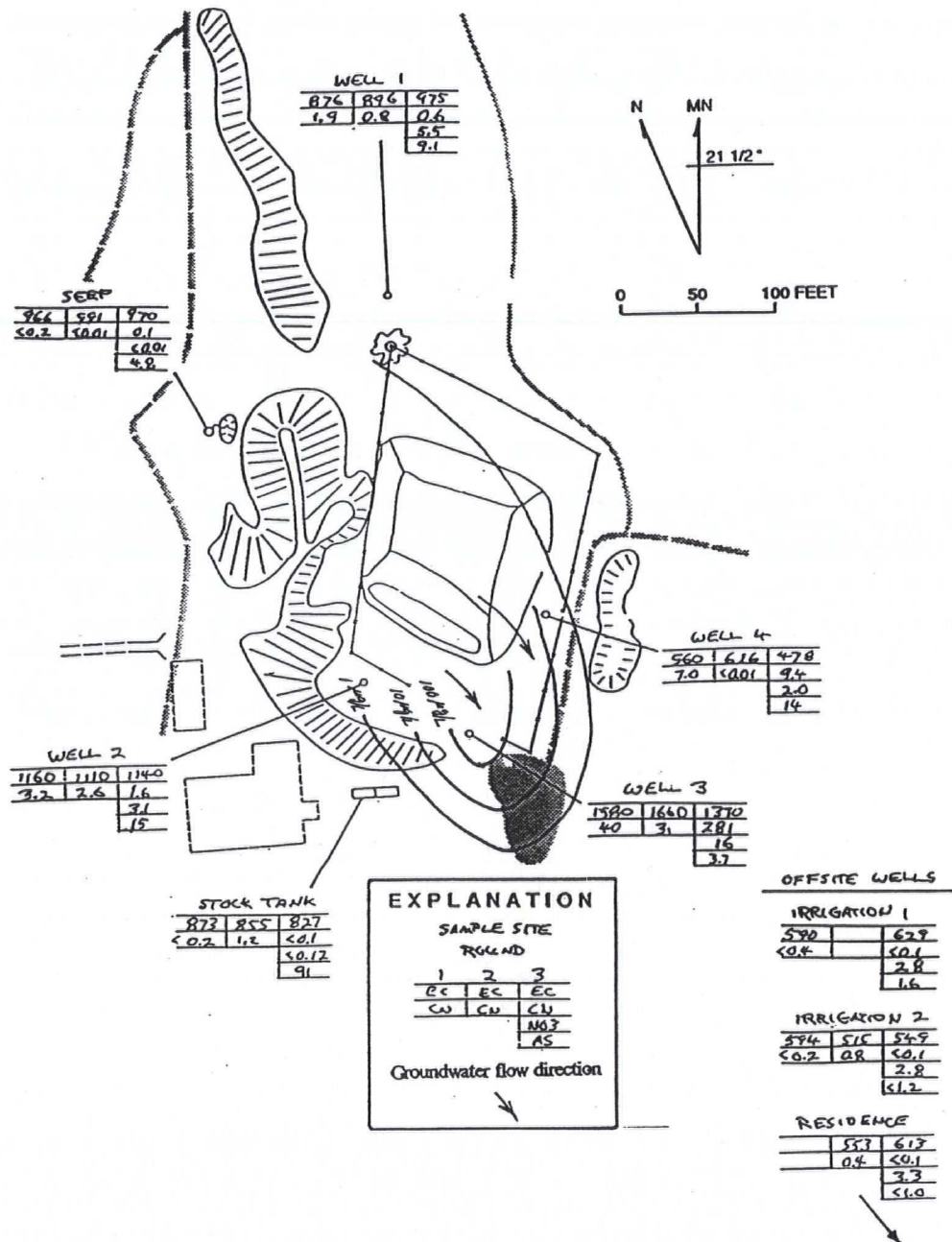


Figure 4.4. The distribution of electrical conductivity (EC in uS), cyanide and arsenic (CN and AS in $\mu\text{g/L}$), and nitrate (NO_3 in mg/L) in ground water and surface water. Electrical conductivity, cyanide, and nitrate values indicate the presence of a dilute plume (shown by inferred contours) extending from the leach heap in the downgradient direction to the southeast. As discussed in the text, other parameters which show a similar distribution include sodium, potassium, and fluoride. The distribution of arsenic, on the other hand, indicates highest values associated with sources to the west of the heap including mine drainage, the mine dump, or bedrock. As discussed in the text, elevated antimony and several other metals also originate west of the heap.

4.4 NATURE OF CONTAMINATION

The analytical results of rock, soil, and water samples and the risk evaluation in Chapters 6 and 7 indicate that two potential contaminants of concern, cyanide and arsenic, merit further discussion, particularly with respect to their amount and chemical form.

4.4.1 Cyanide

4.4.1.1 Cyanide in Rock and Soil

The quantity of cyanide in the leach heap may be estimated by combining measured concentrations with assumptions concerning the central part of the heap where samples were not collected. The distribution of measured concentrations suggests preferential concentrations of cyanide at the toe of the heap, where leachate discharged into the pond. Relatively high concentrations would also be expected next to the liner under the heap materials, since leaching solutions would have flowed through the heap and saturated materials on the liner before travelling toward the pond.

Therefore, for the purpose of estimating quantities, the center was divided into two equal parts. The upper half was assumed to have concentrations of cyanide comparable to that found in the top samples, whereas the lower half was assumed to be comparable to the toe samples. These assumptions should be environmentally conservative, because they probably overestimate the amount of cyanide in the lower half of the heap's center. Table 4.9 lists the data used to derive estimated values for the quantity of contaminants in the heap. The volumes of different parts of the heap are taken from Appendix C, and the concentrations are from Table 4.1.

These data and assumptions yield an average mass of 200 kg (440 lbs) of cyanide, as CN, in the heap. About 2000 kg (4400 lbs) of sodium cyanide (NaCN) is believed to have been applied to the heap during leaching operations (see Section 1.2.2). This amount of NaCN would correspond to 1060 kg of CN. Therefore, on the basis of averaged values, approximately 19% of the cyanide originally applied to the heap remains in place. The remainder of the cyanide either discharged to the leachate pond, where it was removed in metals-laden ore solution or was degraded by treatment or natural processes, or infiltrated into the ground through the liner or through spills.

Table 4.9. List of cyanide and arsenic concentrations and mass in the heap and pond bottom.

HEAP	VOLUME ft3	MASS OF HEAP		CONCENTRATION		MASS OF CONTAMINANT		
		10+6 lbs	10+6 kg	mg/kg RANGE	mg/kg AVERAGE	kg RANGE	kg AVERAGE	lbs AVERAGE
TOTAL CYANIDE								
Top	27720	2.63	1.19	1.7 - 14	5.8	2.0 - 17	6.9	15
Side	43418	4.12	1.87	0.2 - 3.3	1.4	0.4 - 6.2	2.6	5.7
Toe	12144	1.15	0.52	86 - 173	106	45 - 90	55	121
Center	27720	2.63	1.19	86 - 173	106	102 - 206	126	278
Pond	6375	0.61	0.27	5.4 - 101	33	1.5 - 27	8.9	20
Total	117377	11.15	5.06	0.2 - 173	27	150 - 350	200	440
ARSENIC								
Total	111002	10.55	4.78	64 - 650	398	310 - 3100	1900	4200

Mass determined from volumes of different parts of heap (Appendix ?) and density of 95 lbs/ft³.

Toe volume corresponds to south slope in Appendix ?; side volume with north, west, and east slopes; top volume with top half of center block; center volume with bottom half of center block; and pond with 3-foot deep block beneath the pond.

The chemical form of the cyanide may be estimated from data on total and weak acid dissociable cyanide and from correlation with concentrations of other elements found in the heap. Figure 4.5 shows the relation between total and weak acid dissociable cyanide for data from Table 4.1. A linear regression calculation illustrated in Figure 4.5 indicates that about 11 percent of the cyanide is in weak acid dissociable form. As noted in section 4.1.2, some other elements, including sodium, copper, zinc, and lead, tend to be preferentially concentrated at the toe of the heap along with cyanide. Of these elements, sodium would be the most likely to combine with cyanide in a weak acid dissociable form.

The less soluble forms that make up the remaining 89% of the total cyanide may consist of a variety of compounds. The high iron content of the heap material suggests that iron cyanide compounds may predominate in the poorly soluble fraction. Preferential concentration of zinc, copper, and lead at the toe of the heap suggests these elements may also be incorporated into iron cyanide compounds. Chapter 5 discusses contaminant mobility in relation to the probable forms of cyanide.

4.4.1.2 Cyanide in Water

The results of the field and analytical work indicate that a dilute cyanide plume extends from the heap toward Well 3, 25 feet south of the heap in a downgradient direction. The concentration of cyanide in ground water at the site is about 1,000 to 10,000 times lower than the estimated concentration of the original leaching solution. The plume does not extend as far as the nearest water supply well, located 2 miles downgradient in the Horse Springs Coulee aquifer.

Between June and July 1989, cyanide concentrations increased by a factor of about 10 in Well 3. The increase occurred during a period of decreasing water level at Well 3 and rising levels at the other wells. The increase in cyanide was not accompanied by a corresponding increase of the same magnitude in other constituents. The significance of the increase is not known. Perhaps cyanide is held in moderately soluble solid phases in the aquifer material near the heap and is remobilized at low levels by transient infiltration events. The data for weak acid dissociable cyanide indicate that about 10-30 percent of the aqueous cyanide is in the weak acid dissociable form.

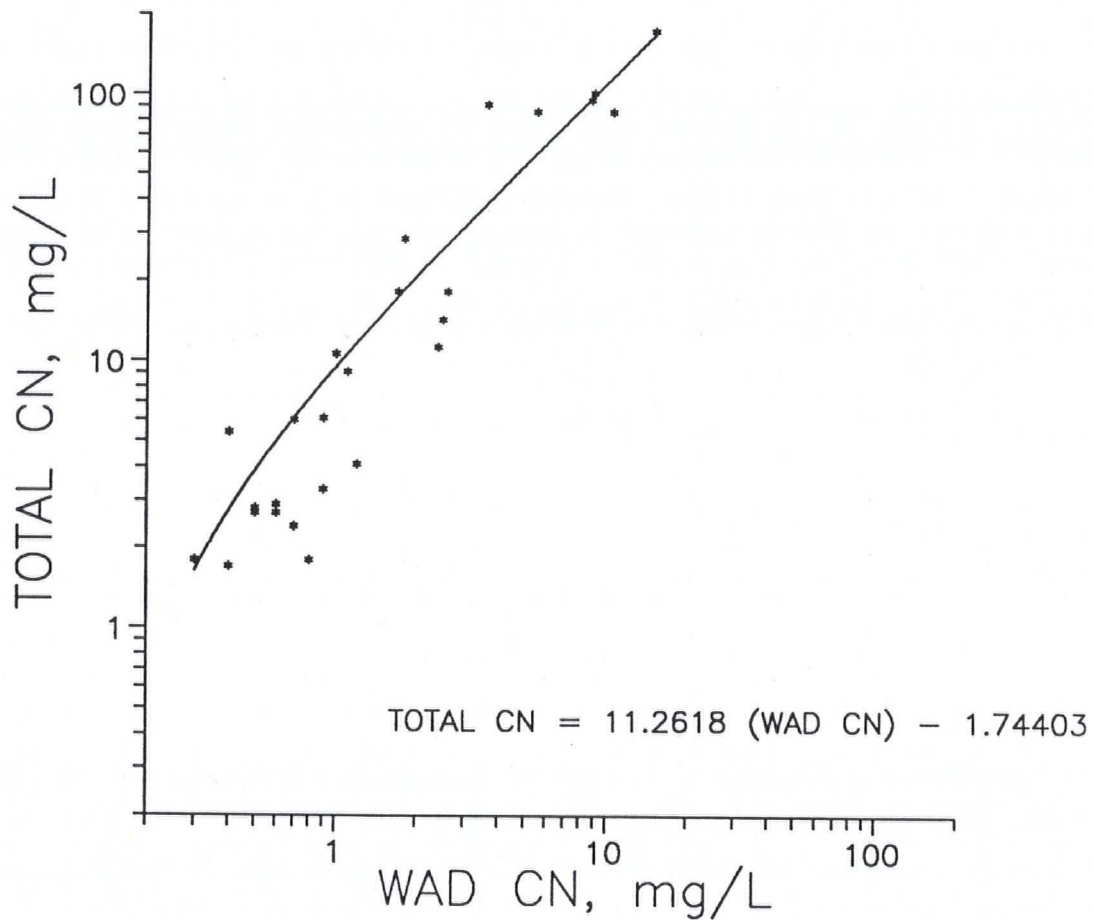


Figure 4.5. Graph showing correlation between total cyanide and weak-acid dissociable (WAD) cyanide in samples of leach heap material. A linear regression curve that accounts for 90% of the residuals is shown as a solid line through the data points.

The relatively high concentration of cyanide in soil beneath the pond liner suggests that at least one likely pathway to ground water is leakage through or over the liner. Concentrations in the soil under the pond liner are comparable to those found at the toe of the heap and probably could only have resulted from direct contact of the soil with leaching solutions. Drainage of leaching solution or runoff into the subsequently plugged well southeast of the heap was considered as another possible pathway. However, samples of this well taken in 1981 and 1983 (Table 1.1) showed cyanide below detection limits (0.002 mg/l) and argue against the well as a contaminant pathway to ground water.

4.4.2 Arsenic

4.4.2.1 Arsenic in Rock and Soil

The quantity of arsenic in the heap may be estimated in a manner similar to that for cyanide, using heap dimensions and measured concentrations (Table 4.9). Unlike cyanide, the arsenic was not preferentially concentrated at the toe or in any other part of the heap. Arsenic quantity can thus be estimated based on the range and average of values for all heap samples. Using the data in Table 4.1, a range of 64 to 650 mg/kg for arsenic in 23 samples, or an average of 398 mg/kg, yields a mass range of 310 to 3100 kg or an average of 1900 kg (4200 lbs) of arsenic in the heap.

The average concentration of arsenic in the leach heap samples was about half of the average concentration in the samples of unleached mine dump material (Figure 4.2). Assuming that the heap material originally contained a similar amount of arsenic to that in the mine dump, perhaps as much as half of the arsenic originally in the heap may have been removed by leaching.

The present capacity of the heap to be a source for arsenic in ground water should be similar to that of the abandoned mine. In 1984, arsenic was detected at an elevated level of 110 $\mu\text{g/l}$ in the leachate pond. This concentration indicates some leaching of arsenic from the heap at that time. A comparable level, up to 91 $\mu\text{g/l}$, occurs in mine drainage at present (Figure 4.4). Currently, the mine drainage contains the highest arsenic values detected in any water at the site.

The mine drainage concentrations and 1984 leachate value may provide an estimate of the general leaching potential of arsenic from bedrock and from mined materials. The potential for arsenic leaching in the future depends on the form in which arsenic

exists in the rock. The form of arsenic was investigated in a petrographic analysis of heap samples (Appendix D). On the basis of electron microprobe results, arsenic probably occurs as sub-microscopic sulfide minerals, including arsenopyrite and arsenic-bearing pyrite. The mobility of arsenic in the sulfide form is discussed in Chapter 5.

4.4.2.2 Arsenic in Water

Arsenic occurs in ground water at concentrations as high as 15 $\mu\text{g/L}$. No clear influence of the heap is apparent in the distribution of the arsenic (Figure 4.4). A comparison of arsenic values for onsite wells with those of wells off site suggests a somewhat higher level of arsenic in ground water at the site than in the main part of the Horse Springs Coulee aquifer. Arsenic in on-site ground water samples ranged from 3.3 to 15 $\mu\text{g/L}$, with an average of 10 $\mu\text{g/L}$ for 4 samples. Offsite samples had values of <1 and 1.6 $\mu\text{g/L}$.

4.4.3 Sources of Contaminants in Ground Water

The distribution of cyanide in ground water, relative to flow direction and to the leach heap, clearly shows that the area of the heap and leachate pond is the source of this contaminant. As discussed under the extent of contaminants in Section 4.3.2.2, several additional parameters, including sodium, potassium, fluoride, nitrate, and nitrite show the same distribution pattern as cyanide for Round 3 samples. The elevated levels of these parameters downgradient of the heap can be inferred to originate from leakage, spillage, or overflow of leaching solutions used on the heap.

As noted above, several additional parameters including arsenic, antimony, barium, chromium, copper, lead, iron, manganese, nickel, silver, and zinc show a divergent distribution pattern with preferentially greater concentrations in Wells 1 or 2. Although present in heap materials, the distribution of these parameters in ground water clearly indicates an origin other than the heap. Wells 1 and 2 are downgradient of three potential sources: the mine dump, mine drainage (well 2 only), and bedrock. The relative influence of these three potential sources is somewhat speculative at present. The proximity of the well intakes to both bedrock and to the probable infiltration path through the mine dump (Figure 3.1b), suggests that either or both could be the primary source for these parameters.

Two of the probable bedrock contaminants, arsenic and antimony, have higher concentrations in well 2 compared to well

1. The elevated levels of arsenic and antimony in well 2 suggest that whatever bedrock or mine dump influence exists for these contaminants is further enhanced by the mine drainage. The high levels of arsenic (Round 3) and antimony (Round 1) in the stock tank also suggest that mine drainage is a potentially major source of these contaminants to ground water.

4.5 SUMMARY

The nature and extent of contamination at Silver Mountain Mine has been evaluated by field geologic mapping, hydrogeologic investigation incorporating four monitoring wells and three offsite water supply wells, and analysis of the chemical composition of 34 samples of leach heap and mine dump material, 20 samples of nearby soils, and three rounds of water samples from seven wells and two surface water sites. Elevated levels of contaminants in solid material are largely confined to mined bedrock that has been crushed through the process of mining and abandoned in piles (mine dump), or that has additionally been abandoned after leaching with cyanide solutions (leach heap). Contaminants are considered elevated in relation to background soils that may be influenced by natural erosion of bedrock and glaciofluvial deposition, but are not influenced by mining activities. Contaminants that appear to be elevated relative to background soils consist primarily of arsenic and other metals and metalloids in the mine dump, and these same constituents plus cyanide in the leach heap. The same contaminants occur at lower, but still elevated, concentrations in shallow soils beneath the heap leach collection pond and in a localized area of shallow soil within 25 feet adjacent to the heap.

Elevated levels of contaminants also occur in ground water at the mine site and in surface water in a stock pond fed by mine drainage. In this respect, elevated aqueous contaminants from the heap are considered elevated relative to concentrations upgradient of the heap. Elevated contaminants from other sources, the mine dump, mine drainage, and bedrock, are considered elevated when more concentrated than the furthest downgradient monitoring well at the site, Well 3. Elevated constituents in ground water consist of cyanide and slightly elevated levels of sodium, potassium, nitrate, nitrite, and fluoride originating from the leach heap, and arsenic, antimony, barium, chromium, copper, chloride, iron, lead, manganese, nickel, silver, and zinc originating either from bedrock or the mine dump. Additionally elevated arsenic and antimony occur in mine drainage.

Ground water contaminants from the leach heap extend in a plume at least as far downgradient as the furthest monitoring well, well 3, 50 feet southeast of the heap. Ground water contaminants from either the mine dump or bedrock are substantially reduced at well 3, which is 100-200 feet downgradient of these potential sources. No ground water contaminants influence offsite water supply wells 2-4 miles downgradient to the southeast of the mine site.

REFERENCES

- Piper, A.M., 1944, A graphic procedure in the geochemical interpretation of water analyses: American Geophysical Union Transactions, v. 25, p. 914-923.
- Russell, Robert H., and Eddy, Paul A., 1972, Geohydrologic evaluation of Aeneas Lake-Horse Springs Coulee, Okanogan County, Washington: Washington Department of Ecology Investigations, January 1972, 16 p., 1 app.

CHAPTER 5 - FATE AND TRANSPORT OF CONTAMINANTS

This chapter examines the processes and rates by which contaminants are transported away from the Silver Mountain Mine site. The emphasis is on the two primary contaminants of concern described in Chapter 4, cyanide and arsenic. For ease of reference and to facilitate computations in this Chapter, metric units following the International System of Units (U.S. Department of Commerce, 1977) are used throughout as the primary units, except where spatial dimensions refer to either maps or well logs drawn in English units. English units are used in the latter cases and in parentheses to the metric units where useful.

5.1 CONCEPTUAL MODEL OF HYDROGEOLOGIC SYSTEM

Quantification of the transport of contaminants at the site was addressed within the framework of a conceptual hydrogeologic model that illustrates the most important physical elements of the site's environment. A hydrogeologic model for Silver Mountain Mine is shown in Figure 5.1 in the form of a block diagram. Six features of the site are illustrated with the transport processes that occur within and between each feature. The features include the bedrock, shallow aquifer, Horse Springs Coulee aquifer, leach heap, leach pond, and mine dump. Transport processes affecting these features by means of water include precipitation, evapotranspiration, runoff, seepage, infiltration, and ground water flow. Additional transport processes in air involving particulates and volatiles are discussed in Chapter 6 for the risk assessment, but are not shown in Figure 5.1.

Examination of Figure 5.1 and consideration of the physical site characteristics and distribution of contaminants described in Chapters 3 and 4 show that the principal transport processes for contaminants from their source areas include the following:

A. Leach Heap

1. Infiltration of precipitation, assuming that the plastic top liner is absent or leaks after a period of aging.
2. Production of leachate by infiltrating precipitation.
3. Movement of leachate from the heap through the unsaturated zone to ground water, assuming that the plastic bottom liner leaks as a result of damage during installation or from aging.

4. Seepage from the toe of the heap to the leachate pond, assuming that the bottom liner and clayey base retard to some extent the downward movement of leachate.

B. Leach Pond

1. Infiltration of leachate to the unsaturated zone and movement to ground water, assuming that either the plastic pond liner leaks or that the pond overflows.

2. Runoff to surface water from pond overflow.

C. Mine Dump

1. Infiltration of precipitation and production of leachate.

2. Infiltration of leachate from the mine dump to the unsaturated zone and movement to ground water.

D. Bedrock

1. Infiltration of precipitation and production of leachate by seepage into the mine workings.

2. Infiltration of mine leachate back into bedrock through the floor of the mine adit.

3. Runoff of mine drainage from the mine opening to surface water.

4. Piping of mine drainage to the stock tank.

5. Infiltration of mine drainage to the unsaturated zone and movement to ground water in the shallow aquifer, as a result of either runoff from the mine or overflow at the stock tank.

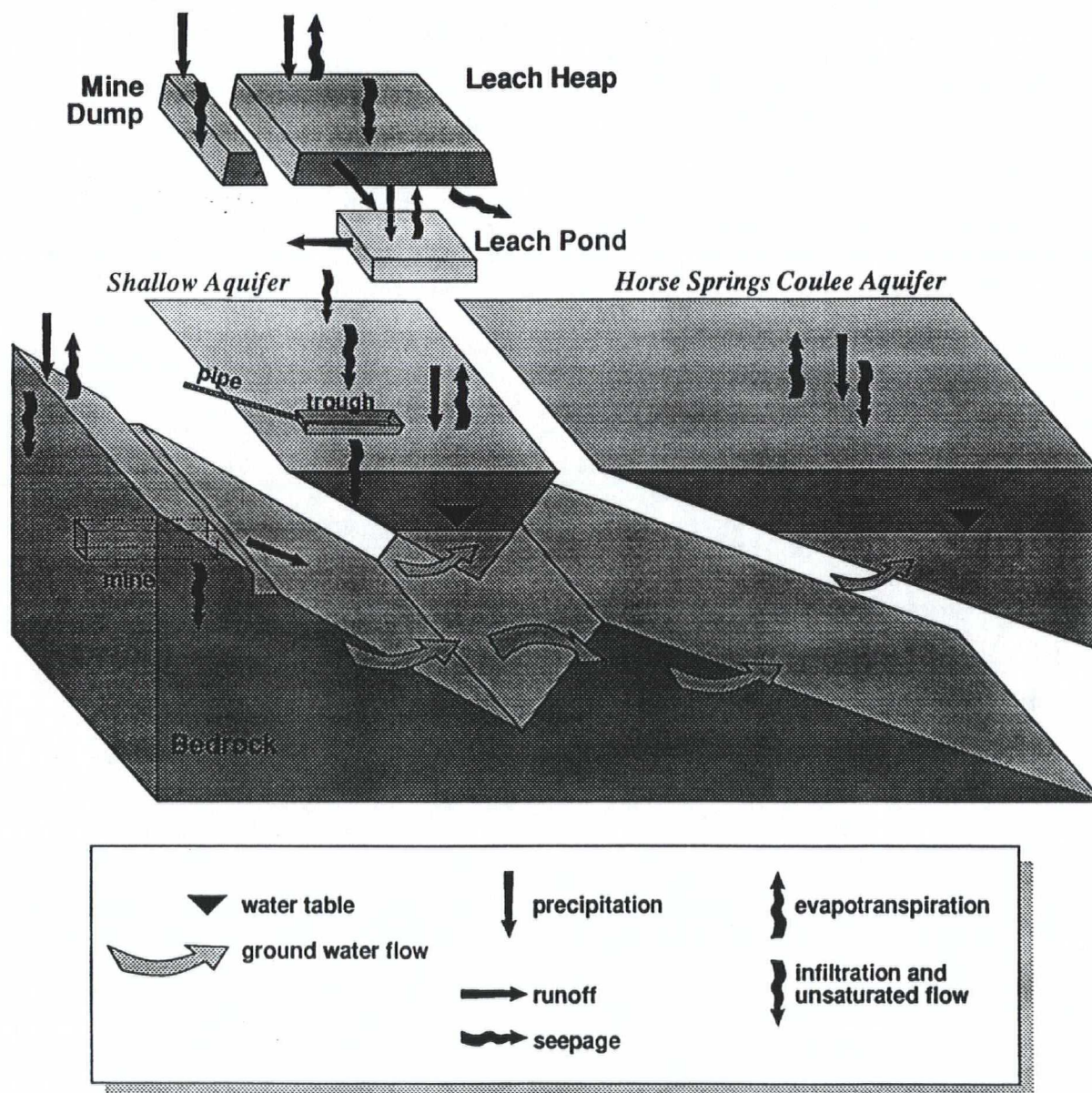


Figure 5.1. Diagram of conceptual model of the hydrogeologic system at Silver Mountain Mine.

The arid nature of the site and lack of surface water suggests that the primary hydrogeologic transport processes of concern are onsite infiltration, movement through the unsaturated zone to ground water, and ground water flow from the shallow aquifer to the main part of the Horse Springs Coulee aquifer. Potential contamination of surface water seems limited to water in the leachate pond and stock tank. If either of these features overflow, the likely path of water is into the unsaturated zone by infiltration.

The transport of contaminants, after infiltration to the shallow aquifer, is with the regional ground water flow. From Figures 3.3 to 3.6, the flow path of ground water is generally to the southeast into the main part of the Horse Springs Coulee Aquifer. Available data indicate that ground water at the site apparently moves from the shallow aquifer to the main part of the coulee through the low ridge of metamorphosed limestone just east of the heap (Figure 3.1). Ground water flow through bedrock is likely to occur preferentially in either surficial fracture zones formed by weathering or in shear zones at depth formed by folding and faulting.

The fate of contaminants during transport is governed by the nature of the contaminant and its reactivity to several transformation processes including chemical speciation, sorption, precipitation and dissolution, hydrolysis, photolysis, volatilization, biodegradation, and bioaccumulation. Fate is evaluated here primarily with respect to mobilization from the solid to the liquid phase and subsequent transport through the unsaturated zone to ground water. First water transport is evaluated, then fate during transport.

5.2 WATER TRANSPORT

5.2.1 Infiltration and Unsaturated Flow

The average linear velocity of infiltrating water is governed by the availability of water at the surface, the hydraulic conductivity, and the moisture content of the unsaturated zone. Several methods were used to place constraints on estimates of the velocity in the unsaturated zone, including infiltration estimates, water balance calculations of the heap, and comparison with literature values of hydraulic conductivity for representative lithologic materials.

The short-term velocity should be greater than the average annual velocity required by the water budget calculation in Chapter 3 of annual recharge to the shallow aquifer. Annual recharge is estimated in Chapter 3 based on 38 cm (15 in) of annual precipitation. The annual recharge estimate was 1140

m³/ha/yr (122,000 gal/ac/yr), which would require on average velocity of at least 0.00031 m/d.

A water balance estimate of discharge rates from the base of the heap was also made using the annual precipitation value partitioned into daily rainfall events that follow seasonal precipitation and evapotranspiration trends. This estimate was made using the water budget modeling code, HELP, of Schroeder and others (1984). Input parameters were based on climatic data for a seasonal distribution of 30 cm (12 in) of annual rainfall concentrated during December through April. Values considered to be most representative with reference to field data were chosen for heap parameters including a heap area of 3200 m² (10,500 ft²), an evaporation depth of 10 cm (4 in), no potential runoff, and the following:

<u>LAYER</u>	<u>THICKNESS</u>	<u>POROSITY</u>	<u>FIELD CAPACITY</u>	<u>HYD COND</u>
Heap	4.3 m	0.52	0.32	0.0002 cm/s
Drain	0.15 cm	0.442	0.256	0.00047 cm/s
Base	0.9 m	0.3647	0.2253	0.000024 cm/s

The simulated discharge from the base of the heap using these parameters and assuming a leaking plastic bottom liner amounted to an average of 0.00014 m/d and a peak daily value of 0.033 m/d.

Maximum estimates of average linear velocity in the unsaturated zone were also made based on well log information of site materials. In Chapter 3, materials in the unsaturated zone are described as glacial sediments consisting primarily of silt, sand, and gravel, capped by silty loam. Maximum saturated hydraulic conductivities for silty sand and gravel generally ranges from 10⁻⁴ to 10⁻¹ cm/s (Freeze and Cherry, 1979). Unsaturated hydraulic conductivities are highly dependent on water content, and may range from these values commonly down to as much as two orders of magnitude less, or 10⁻⁶ to 10⁻³ cm/s. Based on these hydraulic conductivities and an effective porosity of 0.3, velocities may range from 0.0003 to 300 m/d. The lower part of this range would be expected for the Silver Mountain Mine site because of the arid conditions and consequent low water content of soils.

Transport calculations that follow are based on estimates of the most likely range of velocities in the unsaturated zone of 0.003 to 0.3 m/d.

5.2.2 Ground Water Flow

Ground water travels offsite into the main part of the Horse Springs Coulee aquifer where the closest water supply wells lie about 2 miles downgradient. The extent of contaminants beyond the current set of monitoring wells is not known at present but contaminants were not detected in the distant water supply wells. An estimate of potential future effects of onsite ground water on Horse Springs Coulee may be made by comparing the specific discharges for the two areas.

Chapter 3 shows onsite flow directions and provides an estimate of the average linear velocity in the saturated zone of about 0.001 m/d (1.3 ft/yr). The average linear velocity multiplied by the estimated effective porosity yields a specific discharge, or Darcy velocity, of 0.0001 m/d (0.1 ft/yr).

As onsite ground water passes into the main part of Horse Springs coulee, it is entrained in much faster flowing ground water where aquifer materials are more conductive than onsite materials. An estimate of average linear velocity in Horse Springs Coulee is made from data provided by Russell and Eddy (1971). They report the main aquifer materials in Horse Springs Coulee to be composed of coarse-grained recessional glacial outwash. Estimates from Freeze and Cherry (1979) of hydraulic conductivities for these materials are generally 10^{-2} to 10^{-1} cm/s. The hydraulic gradient in the part of the Coulee spanning the area east of the mine site is 0.0097 based on water levels in supply wells T38N/R26E-27Q1 and T37N/R26E-10H1 (Russell and Eddy, 1971, p. 9). The specific discharge based on this gradient and the estimated range of conductivities is 0.084 to 84 m/d.

The ratio of the estimated specific discharges for the shallow aquifer and Horse Springs Coulee aquifer yields values of 800 to 800,000. A reasonable assumption considering the large disparity in flow rates is that complete mixing occurs along flow lines as ground water from low permeability materials at the mine site slowly leaks into high permeability materials in the main part of Horse Springs Coulee. Following this assumption, a conservative value for a dilution factor from the mine site to the main part of the aquifer is about 800.

5.2.3 Uncertainty in Water Transport Estimates

Assumptions and probable data variability inherent in the estimates of velocity and specific discharge result in a very large uncertainty in flow rates. At one extreme, the field measurements of hydraulic gradients are based on water level data that commonly have high precision and accuracy. At the opposite extreme, however, hydraulic conductivity measurements of the

saturated zone by means of slug tests are essentially point measurements having large spatial variability which may not be representative over a large volume of the aquifer. Additionally, hydraulic conductivities of the unsaturated zone and Horse Springs Coulee are estimates that may be inaccurate by a few orders of magnitude. Consequently the primary source of uncertainty is in the values of hydraulic conductivity.

The values used to describe water transport are considered to be the most appropriate based on current knowledge of the site. Refinement of water transport estimates may be made by installing additional test holes and monitoring wells in order to acquire additional data on water levels, hydraulic gradients, and hydraulic conductivities in the saturated zone, and moisture content in the unsaturated zone. Furthermore, infiltration tests could be conducted to better define infiltration rates.

5.3 CHEMICAL TRANSPORT

5.3.1 Contaminants of Concern

Chemical parameters measured at the site include metals and cyanide in heap and mine dump material; metals, cyanide, and organics in adjacent soils; and metals, cyanide, and anions in water. The distribution of elevated levels of these parameters across the site presented in Chapter 4, and estimates of human health risk presented in Chapter 6 indicate that emphasis in evaluation of transport should be placed on two parameters, cyanide and arsenic, with lesser emphasis on nitrate, nitrite, lead, and antimony.

The sources of cyanide and arsenic are substantially different. As described in Chapter 4, cyanide is primarily associated with the leach heap and leachate pond. The distribution of cyanide in ground water indicates a dilute plume of cyanide extends downgradient from the heap.

Arsenic by contrast is associated with not only the leach heap, but also the mine dump and mine drainage. The distribution of arsenic indicates that it occurs naturally in bedrock and is probably present in any mined material at the site. Elevated arsenic also is potentially present in leachate from both mined material and bedrock. Arsenic levels in ground water suggest that slightly elevated concentrations are associated with bedrock. No distinct plume of arsenic can be identified with the leach heap.

An estimate of the potential contaminant loading to ground water can be made by combining the results above for water transport with concentrations and volumes measured in samples

from the heap. From Chapter 4, about 150-350 kg of cyanide and 310-3100 kg arsenic remain in the heap. If heap material was completely leached in a short time period, such as a year, very high levels would impact surface or ground water. If the total masses of the contaminants were incorporated in the estimated yearly infiltration across the area of the heap at a rate of 111 m³/yr (122,000 gal/ac/yr), the resultant concentrations in leachate would be 1400-3200 mg/L of cyanide and 2800-280,000 mg/L of arsenic. More realistic leaching times would be of several years duration because of fate and transport constraints on mobility as water percolates through the source materials and into the unsaturated zone.

The approach used to determine the leachability of materials at the site was to assume a single leach and transport process which would provide the maximum likely leach rate. The basis for the assumptions is a combination of literature information on the fate of cyanide and arsenic coupled with the measured levels of contaminants and estimated hydrogeologic parameters for the site.

5.3.2 Probable Fate of Cyanide

Cyanide compounds and aqueous molecules and ions may be grouped according to their general stability as follows (after Scott, 1984):

	<u>Aqueous Molecule or Ion</u>
1. Free Cyanide	CN ⁻ , HCN
2. Weak Complexes	Zn(CN) ₄ ²⁻ , Cd(CN) ₃ ⁻ , Cd(CN) ₄ ²⁻
3. Moderately Stable Complexes	Cu(CN) ₂ ⁻ , Cu(CN) ₃ ²⁻ , Ni(CN) ₄ ²⁻ , Ag(CN) ₂ ⁻
4. Very Stable Complexes	Fe(CN) ₆ ⁴⁻ , Co(CN) ₆ ⁴⁻ , Au(CN) ₂ ⁻
	<u>Compound</u>
5. Readily Soluble Compounds	NaCN, KCN, Ca(CN) ₂ , Hg(CN) ₂
6. Poorly Soluble Compounds	Zn(CN) ₂ , Cd(CN) ₂ , CuCN, Ni(CN) ₂ , AgCN

The analytical methods described in Chapter 2 determined two types of cyanide, total and weak acid dissociable (WAD), and reported both as concentration of CN. With reference to the above groups, weak acid dissociable cyanide includes primarily groups 1, 2 and much of 3 in water and group 5 in solids. Total cyanide includes all of the groups with the exception of the most stable part of group 4, including cobalt and gold cyanide complexes. The purpose of the two analytical methods is to differentiate between the more mobile and less mobile forms of cyanide. Analytical results in Chapter 4 indicate about 11% of the cyanide in solids and 10-30% in ground water is in the weak acid dissociable form.

The general summary of the fate of cyanide in the environment, as reviewed by Callahan and others (1979), indicates that cyanide ion reacts to form a variety of poorly soluble metal cyanides. However, the highly soluble complex metallocyanides form in water when cyanide ion is in excess. The chemical speciation of cyanide and complexing agents determine the relative amounts of free cyanide and the stable and less stable complexes and compounds. In the heap leach environment, formation of stable metallocyanide complexes are expected in leachate and poorly soluble metallocyanide compounds should occur in the heap.

Volatilization and biodegradation are the dominant processes affecting the stability of free cyanide (Callahan and others, 1979). Below a pH of 8, over 90% of free cyanide exists as HCN and, if open to the atmosphere, will readily volatilize. Biodegradation of cyanide by many organisms has been documented in laboratory experiments, and Callahan and others (1979) conclude that the process is very important at low concentrations in the aqueous environment. Volatilization should occur in leachate that seeps to surface water from the leach heap. The degree of volatilization should decrease as leachate infiltrates into the unsaturated zone. Biodegradation could also occur in bacteria-rich parts of the leach heap and underlying unsaturated zone.

The more stable metallocyanides may be photodecomposed to free cyanide under the action of sunlight (Callahan and others, 1979). This process should be important in leachate that seeps to the ground surface at the edge of the heap, but should not occur after infiltration. Callahan and others (1979) indicate that bioaccumulation and sorption are not important processes in the fate of cyanide.

5.3.2.1 Speciation of Cyanide

Calculations of the projected long-term species distribution

of cyanide in ground water at Silver Mountain Mine were made using an equilibrium speciation model incorporating ion association with thermodynamic equilibrium to determine aqueous species. The model is constrained by mass balance expressions for elements included in analyses of water samples, and by mass action equations for reactions between species. Calculations were made with the geochemical modeling code MINTEQA2 (Brown and Allison, 1987). The thermodynamic data base used for cyanide was compiled by Sehmel (1989) and includes 62 aqueous species. The data base is far more comprehensive than the list above from Scott (1984) and includes a number of species that, if present, occur in extremely minute quantities.

Three assumptions are required for the speciation calculations in order to fix the activity of iron and the redox potential. The assumptions are that the analytical value for iron represents predominantly ferrous iron, ferric iron is fixed by equilibrium with ferric hydroxide ($\text{Fe}(\text{OH})_3$), and redox potential is fixed by the subsequent equilibrium of ferric/ferrous iron. These assumptions are considered to be adequate for a shallow ground water environment.

Table 5.1 lists a representative example of the cyanide species distribution based on the water analysis for well 3, round 3 (Table 4.8-4.9). Under the pH conditions measured for this sample, most of the cyanide occurs as relatively stable iron complexes whereas only about 4% of the total cyanide occurs as free cyanide. The modeled proportion of free cyanide is less than the measured proportion (10-30%) of weak acid dissociable cyanide.

Two important implications relating to the production of HCN may be drawn from the species distribution. First, as long as ground water is not brought to the surface, photodegradation of the stable aqueous complexes should not occur. However with ground water extraction, the low levels of cyanide may be expected to photodecompose to HCN and CN^- .

The second implication derives from the high sensitivity of the species distribution to changes in pH. At the field values of pH 7.3 to 8.2 measured for ground water during the three-month period of sampling from May to July, the calculated species distribution in ground water is very close to the reaction of CN^- to HCN, simply because of the pH. With slight acidification of the water, the calculated abundance of iron cyanide species would be drastically reduced because of a decrease in CN^- , and HCN species would be dominant in water. Under conditions of decreasing pH, the modeled proportion of free cyanide would approach the relatively higher levels of weak acid dissociable cyanide that were measured.

Over the long term, the pH trend in ground water from the site should be toward some lowering of pH from current field values because of the potential for oxidation of sulfides identified in leach and mine dump material (Table 3.1). Furthermore, the distant water supply wells downgradient from the site have somewhat lower pH values measured at 6.9 to 7.6. The conclusion is that, though cyanide in leachate may be largely complexed with iron, the long-term trend should be toward a redistribution of species to free cyanide as leachate seeps through the unsaturated zone, recharges ground water, and moves offsite.

With time, the fraction of weak acid dissociable cyanide should increase from measured values in well 3 of 10-30% as redistribution of species occurs. However, total cyanide should decrease with time because of increased cyanide degradation resulting from volatilization in aerated parts of the unsaturated zone and increased biodegradation.

5.3.2.2 Solubility Limits for Cyanide

The solubility of solid phases was examined with the same geochemical model used for speciation by calculation of saturation indices for various solid cyanide compounds. The saturation index (SI) is the ratio of the activity product for a particular compound and the reaction constant based on thermodynamic data. Positive values of the saturation index indicate oversaturation of a compound; negative values indicate undersaturation. The Sehmel (1989) data base includes thermodynamic information for 27 cyanide compounds.

None of the ground water samples are oversaturated with any cyanide solid phase. Table 5.2 lists the saturation indices for a representative calculation based again on analyses for well 3, round 3. The phase closest to saturation is a potassium zinc ferrocyanide $KZn_{1.5}Fe(CN)_6$. This phase seems to be the most soluble of the cyanide solids for all of the ground water samples.

An estimate of the solubility was made by calculating the amount of cyanide required to saturate the sample under a variety of conditions. At the measured pH conditions for well 3, cyanide saturation would occur at 2.2 mg/L. For comparison, the solubility of sodium cyanide (NaCN), the form of cyanide used in heap leaching, is about 22,000 mg/L. With pH change, the solubility of this phase decreases to minimum of 1.7 mg/L at a pH of 6.5. The conclusion is that current leaching of the pile could potentially produce a few milligrams per liter of cyanide in leachate under present conditions.

Table 5.1. Calculated species distribution for cyanide for the round 3 sample from well 3, July 27, 1989. Distribution calculated with MINTEQA2 geochemical code at pH 7.69, pe 1.47, and 0.281 mg/L of CN. Values listed in order of decreasing concentration. The equilibrated case is for the sample in equilibrium with $KZn_{1.5}Fe(CN)_6$.

SPECIES	SAMPLE			SAMPLE EQUILIBRATED WITH $KZn_{1.5}Fe(CN)_6$		
	CYANIDE CONCENTRATION MOLALITY	$\mu g/L$ as CN	% CN IN SPECIES	CYANIDE CONCENTRATION MOLALITY	$\mu g/L$ as CN	% CN IN SPECIES
CaFe(CN) ₆ -2	8.463E-07	132	47.1	6.981E-06	1090	48.5
Fe(CN) ₆ -4	6.303E-07	98	35.1	5.232E-06	816	36.4
HCN	4.532E-07	12	4.2	6.416E-07	17	0.7
NaFe(CN) ₆ -3	2.306E-07	36	12.8	1.911E-06	298	13.3
Ca ₂ Fe(CN) ₆	7.555E-09			6.201E-08		
KFe(CN) ₆ -3	6.629E-09			6.087E-08		
CN-	6.535E-09			9.254E-09		
Na ₂ Fe(CN) ₆ -2	2.856E-09			2.363E-08		
MgFe(CN) ₆ -2	1.052E-09			8.709E-09		
Ni(CN) ₄ -2	1.594E-10			6.388E-10		
HFe(CN) ₆ -3	1.036E-10			8.593E-10		
Ni(CN) ₂ (aq)	2.621E-11			5.237E-11		
Ni(CN) ₃ -	1.923E-11			5.442E-11		
NaHFe(CN) ₆ -2	6.859E-12			5.678E-11		
K ₂ Fe(CN) ₆ -2	1.381E-12			1.403E-11		
Fe(CN) ₆ -3	8.806E-13			7.094E-12		
CaHFe(CN) ₆ -2	5.620E-13			4.636E-12		
Zn(CN) ₂ (aq)	3.285E-13			8.109E-11		
NiH(CN) ₄ -	2.896E-13			1.160E-12		
MgFe(CN) ₆ -	1.943E-13			1.562E-12		
CaFe(CN) ₆ -	1.416E-13			1.134E-12		
KHFe(CN) ₆ -2	9.376E-14			8.602E-13		
BaFe(CN) ₆ -2	7.308E-14			6.048E-13		
Zn(CN) ₃ -	4.249E-16			1.485E-13		
H ₂ Fe(CN) ₆ -2	2.700E-16			2.236E-15		
NiH ₂ CN ₄ (aq)	2.617E-16			1.048E-15		
BaFe(CN) ₆ -1	1.618E-17			1.300E-16		
NiH ₃ (CN) ₄ +	1.920E-21			7.693E-21		
K ₃ HFe(CN) ₆	5.587E-23			6.283E-22		
Zn(CN) ₄ -2	2.612E-23			1.294E-20		
K ₂ H ₂ Fe(CN) ₆	1.369E-24			1.390E-23		
Fe ₂ (CN) ₆ (aq)	1.418E-29			1.139E-28		

Table 5.2. Calculated saturation indices (SI) for solid cyanide phases in contact with ground water represented by round 3 sample from well 3, July 27, 1989. Solids listed in order of increasing solubility for the unequilibrated sample. Solids closest to saturation are at the top of the list. SI in the second column is for the sample equilibrated with $KZn_{1.5}Fe(CN)_6$.

SOLID PHASE	SI FOR SAMPLE	SI FOR SAMPLE WITH $KZn_{1.5}Fe(CN)_6$
$KZn_{1.5}Fe(CN)_6$	-4.098	0
$Mn_2Fe(CN)_6$	-8.708	-7.791
$Zn_2Fe(CN)_6 \cdot 2H_2O$	-9.393	-4.294
$Cu_2Fe(CN)_6$	-12.352	-11.434
NaCN cri, cub	-12.911	-12.76
KCN i, cub	-13.594	-13.398
$K_2Mn_3Fe(CN)_{12}$	-16.478	-14.555
$K_4Fe(CN)_6 \cdot 3H_2O$	-18.79	-17.695
$K_4Fe(CN)_6$	-19.631	-18.535
$K_3Fe(CN)_6$	-22.141	-21.103
$K_2Ni_3Fe(CN)_{12}$	-22.389	-20.468
$K_4Ni_4Fe(CN)_{12}$	-34.392	-31.466
$Zn_2Fe(CN)_6$	-40.698	-35.599
$K_8Mn_6Fe(CN)_{24}$	-53.027	-48.084
$K_{12}Ni_8Fe(CN)_{36}$	-76.539	-69.592

5.3.3 Probable Fate of Arsenic

The primary form of arsenic in mined material at the site has been identified as sulfide minerals, including arsenopyrite and arsenic-bearing pyrite (Appendix E). The solubility of sulfides is extremely low under reducing conditions. Arsenic-bearing sulfides, however, undergo slow oxidation in surficial materials and in shallow, oxidized ground water. Once oxidation releases arsenic by dissolution of sulfides, arsenic becomes highly mobile in the aqueous environment.

Arsenic fate is largely controlled by ambient conditions of pH and redox potential and the presence of sorbing surfaces. The summary of arsenic fate, as reviewed by Callahan and others (1979), indicates that arsenic strongly sorbs onto clays and iron, manganese, and aluminum hydrous oxides. Sorption in general decreases at both low and high extreme pH values. In water, arsenic occurs mainly in the +3 and +5 oxidation states in an anionic form. The aqueous mobility is affected to some extent by speciation, as the less oxidized form, arsenite, is somewhat more soluble than arsenate. Recent field studies by Moore and others (1988) show that as oxygenated waters high in arsenic and sulfur encounter reducing conditions, sorption loses its importance and sulfide precipitation controls the solubility of arsenic.

Callahan and others (1979) note that volatilization can be important under highly reducing conditions or in the presence of biological activity when arsine gas in the -3 oxidation state or methylated arsenic occurs. Photolysis, however, is not an important process with respect to arsenic.

5.3.3.1 Speciation of Arsenic

Calculations of the likely species distribution of arsenic were made for round 3 samples of ground water and mine drainage using the equilibrium speciation model. The thermodynamic database used for arsenic is derived from the WATEQ3 database (Ball and others, 1981). The database includes 9 aqueous species and 15 solid phases incorporating two oxidation states for arsenic.

The same assumptions used to fix the oxidation state in the cyanide calculations were used for arsenic. That is, ferric/ferrous iron equilibrium was allowed to control redox potential. Arsenic speciation between +3 and +5 was then calculated based on the iron equilibrium. Table 5.3 lists a representative example of the arsenic species distribution using the water analysis for mine drainage, Round 3 (Table 4.8-4.9).

Under the pH conditions measured for this sample, and an iron-controlled Eh of 53 mv, most of the arsenic occurs as arsenate in the oxidized form. Similar results are calculated for ground water. The significance of the speciation calculations is that more extreme reducing conditions, much lower than any encountered at the site in either surface water or shallow ground water, are required to stabilize sulfide phases and provide a long-term sulfidic solubility control for arsenic.

5.3.3.2 Solubility Limits for Arsenic under Oxidizing Conditions

The solubility of oxidized solid phases was also examined for arsenic by calculation of saturation indices. Table 5.4 lists the saturation indices for a representative calculation based again on analyses for mine drainage, Round 3. The only solid arsenic phase showing oversaturation in this water or in calculations for other ground water at the site is barium arsenate, $Ba_3(AsO_4)_2$. Although this phase would severely limit arsenic solubility, it is not commonly found in Nature. None of the water samples are oversaturated with any other arsenic-bearing mineral. The phases closest to saturation are scorodite (hydrated ferric arsenate, $FeAsO_4 \cdot 2H_2O$) and hydrated aluminum arsenate ($AlAsO_4 \cdot 2H_2O$).

An estimate of the solubility was made by calculating the amount of arsenic required to saturate the sample with respect to scorodite. At the measured pH and estimated Eh conditions for mine drainage, arsenic saturation would occur at 109 mg/L. Arsenic saturation by scorodite would occur in Well 1 ground water (Round 3) at 80 mg/L. The lower solubility in Well 1 results from the much higher iron content. The implication of the solubility calculation is any likely solubility controls for arsenic in oxygenated water would allow very high concentrations. However, as discussed below, sorption can provide additional limitation on arsenic concentration.

Table 5.3. Calculated species distribution for arsenic for the round 3 sample of mine drainage from the stock tank, July 27, 1989. Distribution calculated with MINTEQA2 geochemical code at pH 7.87, pe 0.9, and 0.91 mg/L of arsenic. Values listed in order of decreasing concentration. The equilibrated case is for the sample in equilibrium with scorodite, $\text{FeAsO}_4 \cdot 2\text{H}_2\text{O}$.

SAMPLE				SAMPLE EQUILIBRATED WITH $\text{FeAsO}_4 \cdot 2\text{H}_2\text{O}$			
OXIDIZED ARSENIC SPECIES	ARSENIC CONCENTRATION MOLALITY $\mu\text{g/L}$ as AS		% H ₃ AsO ₄ IN SPECIES	ARSENIC CONCENTRATION MOLALITY $\mu\text{g/L}$ as AS	% H ₃ AsO ₄ IN SPECIES		
HAsO ₄ -2	1.149E-06	86	94.8	1.385E-03	104000	94.9	
H ₂ AsO ₄ -	6.310E-08	4.7	5.2	7.369E-05	5500	5.1	
AsO ₄ -3	3.190E-10			4.050E-07			
H ₃ AsO ₄	1.181E-13			1.364E-10			
H ₄ AsO ₃ +	4.244E-19			4.957E-16			
REDUCED ARSENIC SPECIES	ARSENIC CONCENTRATION MOLALITY $\mu\text{g/L}$ as AS		% H ₃ AsO ₃ IN SPECIES	ARSENIC CONCENTRATION MOLALITY $\mu\text{g/L}$ as AS	% H ₃ AsO ₃ IN SPECIES		
H ₃ AsO ₃	5.561E-11	0.004	96.7	6.421E-08	4.8	96.7	
H ₂ AsO ₃ -	1.897E-12	0.0001	3.3	2.215E-09	0.2	3.3	
HAsO ₃ -2	1.046E-16			1.261E-13			
AsO ₃ -3	3.997E-22			5.075E-19			

Table 5.4. Calculated saturation indices (SI) for solid cyanide phases in contact with mine drainage represented by the round 3 sample, July 27, 1989. SI listed in order of increasing solubility for the unequilibrated sample. Solids closest to saturation are at the top of the list. SI in the second column is for the sample equilibrated with scorodite, $\text{FeAsO}_4 \cdot 2\text{H}_2\text{O}$.

OXIDIZED ARSENIC PHASE	SI FOR SAMPLE	SI FOR SAMPLE WITH $\text{FeAsO}_4 \cdot 2\text{H}_2\text{O}$
ALASO ₄ .2W	-7.958	-4.897
FEASO ₄ .2W	-8.435	0
CA3 (ASO ₄) 26W	-10.748	-4.658
MN3ASO ₄ 28W	-14.401	-8.313
ZN3ASO ₄ 22.5W	-15.118	-9.005
AS2O5	-32.688	-26.562

REDUCED ARSENIC
PHASE

ARSENOLITE	-37.847	-25.595
CLAUDETITE	-37.609	-25.358

5.3.4 Transport of Cyanide and Arsenic

A combination of the above estimates of velocity and source leachate concentrations allows evaluation of the likely concentration of cyanide and arsenic that would enter ground water under present conditions. The one-dimensional solute transport equation was used to estimate the concentrations of contaminants after traveling from the ground surface through the unsaturated zone to ground water. The transport equation is:

$$D(\partial^2 C / \partial x^2) - v(\partial C / \partial x) - \lambda RC = R(\partial C / \partial t)$$

where C = contaminant concentration
 v = average linear velocity
 x = travel distance through unsaturated zone
 t = travel time
 R = retardation coefficient
 λ = degradation coefficient

 D = dispersion coefficient
 $= \alpha v$

 α = dispersivity

The solution of the transport equation by van Genuchten and Alves (1982) was used in the form presented by Mills and others (1985, p. 408).

Calculations used a velocity range of 0.003-0.3 m/d as estimated in section 5.2.1; a dispersivity of 0.06 m suggested by various field data compiled by Gelhar and others (1985) for unsaturated media; a travel distance of 6 m from the ground surface to the water table (table 3.3); and travel times of 1 day, 1 year, and 7 years.

5.3.4.1 Cyanide Degradation

An overall degradation rate was assumed to occur for cyanide to incorporate the fate processes of volatilization in aerated parts of the unsaturated zone and biodegradation. An estimate of

degradation under site conditions was based on historical data on cyanide in leachate. Leachate concentrations of samples collected from 1981 through 1984 (Chapter 1) by the Washington Department of Ecology (WDOE) were evaluated to determine the rate of reduction in cyanide in leachate during the period when the heap was uncovered. The historical data were screened to remove analyses of samples that closely followed the pond treatment efforts of the Department of Ecology as these would not have been representative of natural degradation processes. The remaining leachate data are considered to be affected directly by natural degradation of the heap and leachate, and indirectly by partial removal of cyanide from the heap during rinsing in 1983. Screened concentrations are as follows:

<u>DATE</u>	<u>CN, mg/L</u>
June 11, 1981	1100
October 26, 1981	430
November 19, 1981	600
April 5, 1982	220
December 3, 1982	30
November 29, 1983	9.2
September 4, 1984	7.6
November 7, 1984	2.3

These values are plotted in Figure 5.2. An exponential function fitted to the data yields a coefficient of $-0.00465/\text{day}$. The coefficient describes the reduction of cyanide in natural leachate that percolated through the heap under ambient weather conditions during the 4 1/2-year period. This coefficient was used as an estimate of cyanide degradation in application of the solute transport equation.

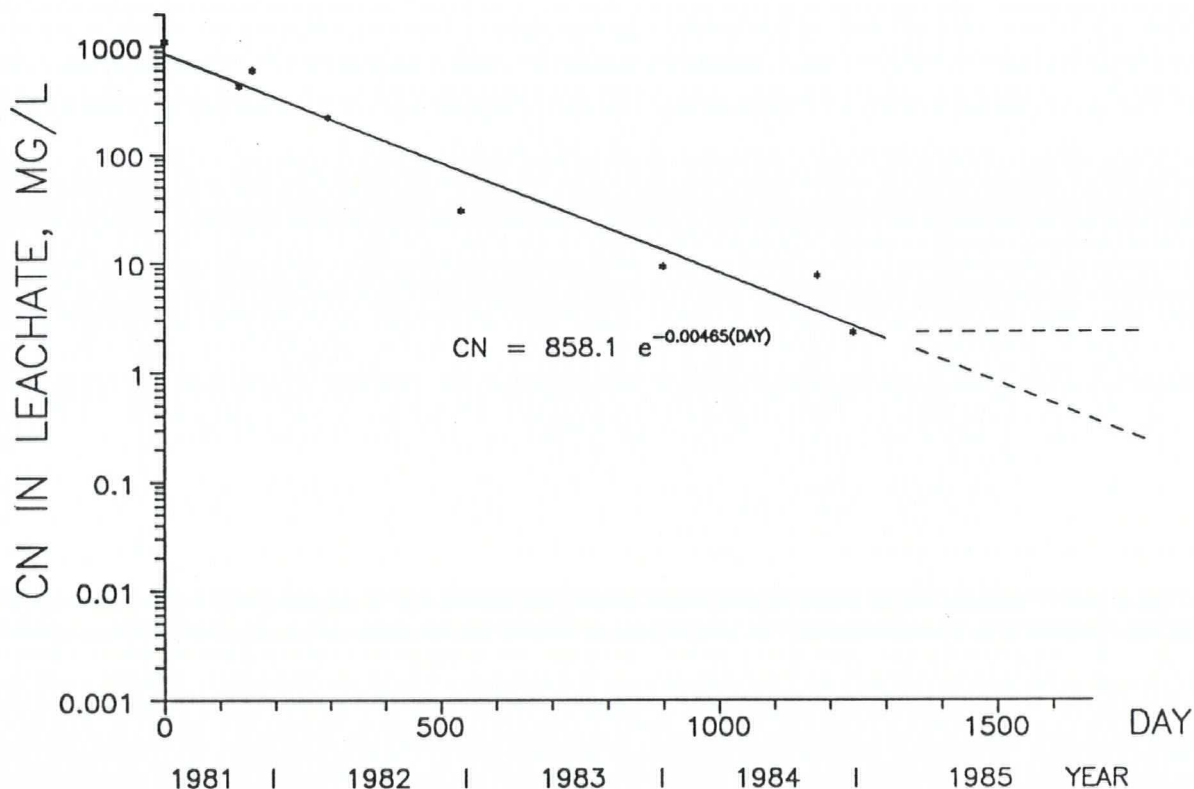


Figure 5.2. Decrease in cyanide concentration in leachate during the post-operation period from 1981 to 1984. Exponential expression is a best-fit curve, where CN is cyanide concentration in mg/L. Day is the day number beginning with the data point on June 11, 1981, which is the time of the first sample following the end of operations. On this scale, January 1, 1990, would be Day 3123. The dashed lines represent two scenarios of projected leachate concentrations. The upper dashed line represents no further degradation beyond 2.3 mg/L reached in November 11, 1984. The lower dashed line represents continuing degradation at the earlier rate.

5.3.4.2 Arsenic Adsorption

Sorption was assumed to occur for arsenic in order to allow for the attenuation effect of site materials rich in iron and aluminum sorbant surfaces. An estimate of a sorption partition coefficient was made by examining literature values for arsenic in other settings, as well as EPA data on arsenic sorption during batch experiments conducted for another mining site in EPA Region 10.

In a review of sorption test procedures, Roy and others (1987) present arsenate sorption data for a silt loam soil on glacial drift in Illinois. These data yield partition coefficient values (K_d) of about 4 mL/g at arsenic concentrations in the region of 30-80 mg/L. Batch test data for clayey soils from a proposed mine site in Idaho (EPA Region 10 files) produced partition coefficient values of 7-70 mL/g for arsenic concentrations of less than 1 mg/L.

A range of 4-70 mL/g for an arsenic partition coefficient would result in initial source concentrations of 1.1-20 mg/L in ground water recharge near the heap and 1.5-27 mg/L in mine drainage, provided that sorption is assumed to limit the concentrations based on ferric arsenate solubility. In the solute transport calculations for arsenic, sorption was assumed to occur in the source material, rather than in the unsaturated zone below the source. Retardation coefficients were not calculated for the transport path because the primary objective for the transport calculation was long-term steady-state concentration.

5.3.4.3 Projected Ground Water Concentrations

Transport calculations were made to determine the steady-state concentration ratio C/C_0 after movement of contaminants through 6 m of unsaturated zone to the water table. C represents concentration at the water table; C_0 represents the initial source concentration during infiltration. The results of the transport calculation are presented below for cases including degradation of cyanide and sorption of arsenic:

		<u>AVERAGE LINEAR VELOCITY</u>						<u>TIME</u>
		<u>0.003 m/d</u>		<u>0.3 m/d</u>		<u>300 m/d</u>		
		CN	As	CN	As	CN	As	
C/C_0	0	0	0	0	0	0.99	1	1 d
	0	0	0.91	1	0.99	1	1	1 yr
	0.00013	1	0.91	1	0.99	1	1	7 yr

From these results, if the initial cyanide leachate concentration were 2.2 mg/L as indicated in section 5.3.2.2, recharge to ground water would be degraded to a steady state concentration of about 2 mg/L after 1 year and a moderately fast average linear velocity of 0.3 m/d. Cyanide would be degraded to 0.0003 mg/L after 7 years and the likely more correct slower velocity of 0.003 m/d. The low concentration of projected cyanide relative to current maximum concentrations in ground water of 0.28 mg/L in well 3 suggest that all cyanide currently in ground water infiltrated several years ago when the pile was subject to heap leaching and surface concentrations were greater than at present.

Steady state arsenic concentrations are projected to remain the same as at infiltration, which from section 5.3.4.2 could be as great as 1.5 to 27 mg/L. A basic assumption for such high levels in leachate is that relatively complete oxidation of sulfide minerals occurs at the ground surface or in shallow depths in the aquifer. Sulfide oxidation rates were not estimated but are likely to be low because of the arid environment. The assumption of complete sulfide oxidation represents an environmentally conservative approach to estimating arsenic concentration. Current levels of arsenic in water indicate that preferential oxidation of sulfides and arsenic release is primarily associated with the mine workings, rather than with the leach heap or mine dump.

Contaminants in the shallow aquifer should undergo dilution during flow into the main part of Horse Springs Coulee. The conservative dilution estimate of 800 from section 5.2.2 indicates that most trace constituents, including cyanide, would be diluted to values below those currently found in offsite water supply wells. Arsenic is a potential exception. If concentrations of arsenic as high as 27 mg/L recharged the shallow aquifer, the projected dilution could allow levels as high as 0.03 mg/L to move into the Horse Springs Coulee aquifer. In contrast, samples from downgradient water supply wells contained no more than 0.002 mg/L.

5.3.5 Uncertainty in Contaminant Transport Estimates

Uncertainty associated with the water transport estimates (section 5.2.3) and assumptions used in the estimation of leachate concentrations, cyanide degradation, and arsenic sorption place high uncertainty on the projected steady state concentrations of contaminants in ground water.

Leachate concentrations are based in part on equilibrium thermodynamic modeling. The adequacy of the model is only as good as the database. If important species are absent from the

database or not included in sample analyses, significant errors may occur in solubility estimates. The MINTEQ database is one of the most extensive thermodynamic compilations available and should be adequate for the parameters of concern here.

Assumptions on redox potential are based on iron equilibria. This assumption may be adequate for the near surface environment at Silver Mountain Mine, but is probably not representative of conditions at depth in ground water or in any localized area that is highly enriched in sulfides. The effect of the assumption depends on the parameter of interest, particularly those that undergo redox reactions such as arsenic. In a sulfur-rich environment such as the mine site, the effect of reducing conditions on the assumption of iron equilibrium should be to overestimate the solubility of arsenic. In a low-sulfur environment such as Horse Springs Coulee, the effect of reducing conditions is to underestimate the solubility of arsenic. The conservative approach used above for estimating arsenic in leachate should account for errors in the redox assumption.

Overall thermodynamic equilibrium may not occur in the low temperature, near-surface environment because of biological activity and slow reaction rates. However, the concept of using the affinity for reactions to tend toward equilibrium is sound. The error in the approach used here would likely be to overestimate contaminant concentrations because slow reaction rates should tend to reduce the amount of precipitates and hence solubility controls for contaminants.

The degradation rate for cyanide cannot be refined without additional site-specific data. The rates used here are thought to be appropriate for this site. The nine-year history for the site and the relatively low levels of cyanide in ground water suggest that the rates are not unreasonably high or low.

The adsorption coefficient for arsenic also cannot be refined without site-specific data. In general, the leachate evaluation is probably environmentally conservative. Ground water concentrations greater than 30 mg/L are uncommon except in areas of disposal of oxidized arsenic wastes from industrial activity.

5.4 SUMMARY

The fate and transport of cyanide and arsenic in water from the Silver Mountain Mine site has been evaluated through estimates of average linear velocity in the unsaturated zone and in ground water, speciation and solubility limits, and degradation and sorption.

The future impact of cyanide on ground and surface water is primarily controlled by the amount and form of cyanide remaining in the heap, and by velocity and degradation rates. Measurements of total and weak acid dissociable cyanide indicate that the cyanide in heap material is mostly in the form of poorly soluble iron cyanide compounds. The estimated velocity and degradation rates for cyanide indicate that the levels now occurring in ground water probably originated in spillage or leachate pond overflow at the time of leach heap operations during 1980 and 1981. Probably little, if any, leachate has been produced since the heap was covered in 1984. However, with time and deterioration of the plastic top and bottom liners, leaching of cyanide from the heap would be expected to resume with transport of cyanide to ground water.

Infiltration of cyanide is projected to occur at progressively reduced concentrations and rates as a result of degradation, including speciation to hydrogen cyanide (HCN) and subsequent volatilization. Projected maximum concentrations of cyanide in leachate are on the order of a few milligrams per liter. Infiltration of leachate at these concentrations is projected to degrade to significantly lower levels during passage through the unsaturated zone.

The future impact of arsenic on ground and surface water is primarily controlled by the amount and form of arsenic in all mined materials, including the heap and mine dump, and in bedrock, and by the sorption capacity of iron- and aluminum-rich soils. The estimated solubility of arsenic and sorption capacity of soils indicate that as the surficial piles oxidize, leachate from the heap and the mine dump could produce high concentrations of arsenic on the order of a few tens of milligrams per liter. Retardation of initially high concentrations of arsenic in leachate could occur during infiltration. However, with time and saturation of sorption sites, arsenic levels impacting ground water could reach the same levels of arsenic as infiltrating leachate. Current levels of arsenic in ground water indicate that oxidation of the mine dump and buried bedrock has not yet progressed to the point of producing highly concentrated leachate. Current elevated levels of arsenic in mine drainage, however, indicate that oxidation may be taking place in the mine workings and that a potential exists for higher concentrations in leachate in the future from any of the mined material.

No impact of contaminated ground water from the mine site exists at present at water supply wells in the main part of Horse Springs Coulee aquifer. The projected impact from estimated future levels of contaminants is significantly less in the Horse Springs Coulee aquifer than in the shallow aquifer at the mine site because of dilution resulting from a large contrast in ground water flow between the two areas.

REFERENCES

- Ball, J.W., Jenne, E.A., and Cantrell, M.W., 1981, WATEQ3: a geochemical model with uranium added: U.S. Geological Survey, Open-File Report 81-1183, 80 p.
- Brown, David S. and Allison, Jerry D., 1987, MINTEQA1, an equilibrium speciation model: user's manual: U.S. Environmental Protection Agency EPA/600/3-87/012, 92 p.
- Callahan, Michael A., Slimak, Michael W., and others, 1979, Water-related environmental fate of 129 priority pollutants: U.S. Environmental Protection Agency, volume 1, EPA-440/4-79-029a, 514 p.
- Freeze, R. Allan and Cherry, John A., 1979, Groundwater: New Jersey, Prentice-Hall, Inc., 604 p.
- Gelhar, L.W., Mantaglou, A, Welty, C., and Rahfeldt, K.R., 1985, A review of field-scale physical solute transport processes in saturated and unsaturated porous media: Electric Power Research Institute, EPRI EA-4190.
- Hoye, Robert, 1987, Gold/silver heap leaching and management practices that minimize the potential for cyanide releases: U.S. Environmental Protection Agency, EPA/600/2-88-002, 103 p.
- Mills, W.B., Porcella, D.B., Unga, M.J., and others, 1985, Water quality assessment: a screening procedure for toxic and conventional pollutants, part II: U.S. Environmental Protection Agency, EPA/600/6-85/002b, 444 p., 9 app.
- Moore, Johnnie N., Ficklin, Walter H., Johns, Carolyn, 1988, Partitioning of arsenic and metals in reducing sulfidic sediments: Environmental Science and Technology, vol. 22, p. 432-437.
- Mudder, T.I. (ed.), 1989, The chemistry, analysis, toxicity, and treatment of cyanidation wastewaters: Bellevue, Washington, Steffen, Robertson, and Kirsten.
- Roy, W.R., Krapac, I.G., Chou, S.F.J., and Griffen, R.A., 1987, Batch-type adsorption procedures for estimating soil attenuation of chemicals: U.S. Environmental Protection Agency, EPA/530-SW-87-006, 183 p.

- Schroeder, P.R., Morgan, J.M., Walski, T.M., and Gibson, A.C., 1984, The hydrologic evaluation of landfill performance (HELP) model: U.S. Environmental Protection Agency Technical Resource Document, volume 1, EPA/530-SW-84-009, 120 p., and volume 2, EPA/530-SW-85-010, 256 p.
- Semel, G.A., 1989, Cyanide and antimony thermodynamic database for the aqueous species and solids for the EPA-MINTEQ geochemical code: Richland, Pacific Northwest Laboratory, PNL-6835, 63 p., 6 app.
- U.S. Department of Commerce, 1977, The International System of Units (SI): National Bureau of Standards NBS Special Publication 330, 1977 edition, 41 p.
- Van Genuchten, M. Th. and Alves, W.J., 1982, Analytical solutions of the one-dimensional convective-dispersive solute transport equation: U.S. Department of Agriculture Technical Bulletin 1661, 149 p.

Chapter 6 - HUMAN HEALTH RISK ASSESSMENT

This chapter describes the baseline human health risk assessment for Silver Mountain Mine. The assessment identifies risks which potentially could occur from exposure to contaminants at the site assuming there is no action to clean up the site or prevent exposure.

6.1 CONTAMINANT IDENTIFICATION

As described in Chapters 4 and 5, inorganic contaminants are the most prevalent at Silver Mountain Mine. Sampling of soils and water described in Chapter 4 quantified the extent of inorganic contamination in these media. Soils sampling conducted in the areas most likely to be contaminated with organic chemicals (eg. fuel, solvents, etc.) identified few of the target organic contaminants listed in Table 2.3. From Chapter 4, the low levels of methylene chloride, benzoic acid, bis (2 ethylhexyl) phthalate, benzyl alcohol and acetone found in some samples are believed to be laboratory contaminants. As a result, these are not discussed further in the risk assessment. A number of compounds not listed in Table 2.3 were tentatively identified. These have not been included in the assessment because they were only found at low concentrations (up to 8.4 mg/kg), and they have not been positively identified.

Another group of chemicals not included in the assessment are sodium, potassium, calcium, magnesium and iron. These chemicals are considered to be essential nutrients, with no known toxicity at concentrations found at Silver Mountain Mine. The toxicology of these chemicals is discussed further in Appendix L.

Aluminum is also not included in the assessment though recently increasing attention has focused on the potential link between aluminum exposure and presenile and other dementias. At this time a cause/effect relationship between aluminum exposure and illnesses such as Alzheimer's disease is only speculative, and toxicity reference values have not been developed. Aluminum concentrations in soil and water are not elevated at the Silver Mountain Mine site (Chapter 4) and a person's normal 30-50 mg/day dietary intake of aluminum (Bjorksten, 1982) is expected to far exceed that from site related contamination. For these reasons aluminum has not been evaluated in the risk assessment. Appendix L provides further discussion of its toxicity.

Table 6.1 lists the remaining inorganic chemicals analyzed in soil and water, all of which were detected in both media.

Table 6.1. Noncarcinogenic Effects and Reference Values.

Compound	Chronic oral RfD	Rfd data source	Critical effects	Uncertainty factor	Modifying factor	Level of confidence
Antimony	4.0E-04	IRIS	longevity, blood glucose, cholesterol level	1000	1	Low
Arsenic (1)	1.0E-03	(1)	nerve, skin, liver, GI			
Barium	5.0E-02	HEAST	nervous system	100	1	Medium
Beryllium	5.0E-03	IRIS	skin, lung	100	1	Low
Cadmium	5.0E-04	HEAST	renal damage	10		
Chromium VI	5.0E-03	IRIS	kidney, skin	500	1	Low
Chromium III	1.0E+00	IRIS	kidney, skin	100	10	Low
Cobalt	Not available	IRIS/HEAST	polycythemia, thyroid, cardiac, bone marrow			
Copper (2)	3.8E-02	(2)	gastrointestinal disorders	2		
Cyanides						
Cyanide, free	2.0E-02	IRIS	weight loss, thyroid effects, myelin degeneration	100	5	Medium
Cyanogen	4.0E-02	IRIS	" " " "	100	5	Medium
Cyanogen bromide	9.0E-02	IRIS	" " " "	100	5	Low
Barium cyanide	7.0E-02	IRIS	hypertension	100	1	Low
Calcium cyanide	4.0E-02	IRIS	weight loss, thyroid effects, myelin degeneration	100	5	Medium
Chlorine cyanide	5.0E-02	IRIS	" " " "	100	5	Medium
Copper cyanide	5.0E-03	IRIS	decreased body/liver weight, liver effects	1000	1	Medium
Hydrogen cyanide	2.0E-02	IRIS	weight loss, thyroid effects, myelin degeneration	100	5	Medium
Potassium cyanide	5.0E-02	IRIS	" " " "	100	5	Medium
Potassium Ag cyanide	2.0E-01	IRIS	" " " "	100	5	Medium
Sodium cyanide	4.0E-02	IRIS	" " " "	100	5	Medium
Zinc cyanide	5.0E-02	IRIS	" " " "	100	5	Medium

Notes:

(1) Agency for Toxic Substances and Disease Registry. Toxicological Profile for Arsenic. ATSDR/TP-88/02. 3/89. The RfD for arsenic is currently under review by EPA. The value given is the previously verified RfD (now withdrawn) which is supported by the ATSDR Toxicological Profile.

(2) Drinking Water Criteria Document for Copper. USEPA. Final Draft, Office of Drinking Water. EPA/600/x-84-190-1. 3/85. The reference value given is for acute exposure to an adult.

IRIS: Integrated Risk Information Service, USEPA Database.
7/89 online search.

HEAST: Health Effects Summary Tables. USEPA. Third Quarter FY 1989.

Table 6.1. Noncarcinogenic Effects and Reference Values. (Continued)

Compound	Chronic oral Rfd	Rfd data source	Critical effects	Uncertainty factor	Modifying factor	Level of confidence
Fluoride	6.0E-02	IRIS	dental/skeletal fluorosis	1	1	High
Lead	Not available	IRIS/HEAST	anemia, CNS, kidney			
Manganese	2.0E-01	HEAST	CNS, liver	100		
Mercury						
inorganic	3.0E-04	HEAST	kidney	1000		
methyl	3.0E-04	HEAST	CNS	10		
Molybdenum	Not available	IRIS/HEAST	liver and kidney degeneration			
Nickel (nickel soluble salts)	2.0E-02	IRIS	contact dermatitis, non-specific	100	3	Medium
Nitrate (3)	1.0E+00 (as N) 4.5E-00 (as NO3)	IRIS	methemoglobinemia (infants)	1	1	High
Nitrite (3)	1.0E-01 (as N) 3.3E-01 (as NO2)	IRIS	methemoglobinemia (infants)	1	1	High
Selenium (selenious acid)	3.0E-03	IRIS	skin, tooth mottling, GI distress	10	1.5	High
Silver	3.0E-03	IRIS	argyria	2	1	Medium
Thallium (thallic sulfate)	8.0E-05	IRIS	alopecia, enzyme effects	3000	1	Low
Tin and compounds	6.0E-01	HEAST	immunotoxicity	100		
Vanadium	9.0E-03	IRIS	gastrointestinal, enzyme effects	100	1	Low
Zinc	2.0E-01	HEAST	antagonizes Cd and Pb effects	10		

IRIS: Integrated Risk Information Service, USEPA Database. 7/89 online search.

HEAST: Health Effects Summary Tables. USEPA. Third Quarter FY 1989.

(3) Reference values for nitrate and nitrite are applied for subchronic or acute exposures since the critical effect is methemoglobinemia in infants.

Table 6.1 also indicates whether noncancer toxicity reference values are available for each chemical. Table 6.2 lists chemicals identified at the site which have been classified as carcinogens, their weight of evidence classification and their cancer reference values (slope factors) if available. To the extent possible the risk assessment has included a quantitative evaluation of all chemicals found at the site which have toxicity reference values. Chemicals for which no reference values are available are discussed qualitatively in the uncertainty section.

6.2 TOXICITY ASSESSMENT

The toxicity assessment is intended to provide information on the effects which may be caused by a chemical, and the doses at which these effects occur, or dose response assessment.

Information to determine the potential for a contaminant to cause adverse health effects may be obtained from three types of studies; human studies, animal studies, and supporting information, such as in-vitro bioassays. Generally, properly conducted human epidemiologic studies which show a positive relationship between exposure to a substance and an adverse effect provide the most convincing evidence of human risk. Unfortunately there are few human studies which can be used to estimate risk from environmental exposure (such as at Silver Mountain Mine) because exposure conditions and populations are often not comparable, exposure is often to a mixture rather than a single chemical, and many studies do not have the ability to detect adverse effects which might occur from long term exposure to low contaminant concentrations. If adequate human data are available, they are weighted more heavily than animal or supporting data in dose response assessment.

For most chemicals, good human data are not available and animal data must be used to evaluate potential effects and dose response. In general EPA believes that it is valid to use information from properly conducted animal studies to predict risks which might occur in humans. However, due to the differences in metabolism, pharmacokinetics, etc. between animals and humans, and between different animal species, there is uncertainty in predicting human risk from animal studies.

Other information, such as metabolic studies, in-vitro mutagenic bioassays, and structure-activity relationships can be used to support conclusions drawn from human or animal studies. This is especially useful in determining the weight of evidence for cancer classification. Only under very limited circumstances is this type of information used to make dose response predictions, since of the three data types, it is the furthest removed from actually measuring human risk.

Summary information regarding the critical effects and dose response reference values for Silver Mountain Mine contaminants is presented here. For detailed discussions of chemical specific toxicity, the reader should refer to Appendix L. Arsenic, lead, nitrate/nitrite, and copper have been discussed separately in the text due to complexity in assessing their risk.

6.2.1 Noncarcinogenic Effects

Adverse human health effects may be grouped into two broad categories; carcinogenic and noncarcinogenic. Noncarcinogenic effects are any effects other than cancer that can be caused by acute (< 2 weeks), subchronic (2 weeks - 7 years), or chronic (> 7 years) exposure to a substance (USEPA, 1989p). Examples of such effects include liver disorders, anemia, neurologic degeneration, teratogenicity, behavioral disorders, etc. The body responds in a variety of ways minimize adverse effects from chemical exposure. Physiologic, metabolic and other mechanisms such as bone and lipid storage, and the glutathione and cytochrome P450 enzyme systems, may provide initial protection against adverse effects. However, there is a dose or threshold at which these protective mechanisms are overcome and the toxic effect is manifested. Based on the threshold concept, EPA has established a procedure to review human, animal and other toxicity studies and establish chemical specific reference doses (RfDs). These are doses, in terms of milligrams of contaminant per kilogram of body weight per day, at which it is expected there will be no adverse (noncarcinogenic) effect.

When establishing an Rfd, EPA first identifies the most critical study. From that study the highest dose which produces no adverse effect (no observed adverse effect level; NOAEL) or the lowest dose which produces an adverse effect (lowest observed adverse effect level; LOAEL) is determined. Uncertainty factors of from 1 to 10,000 are used to adjust this dose for differences between animals and humans, and to adjust for the strength and type of evidence in the study. A modifying factor (1-10) can also be applied if there are other reasons for adjusting (lowering) the reference dose; for example if the overall confidence in the toxicity database is low.

The EPA Rfd Workgroup has been formed to establish and review Rfd's. If the workgroup develops or reviews and verifies a reference dose, a statement is made regarding the overall confidence (high, medium or low) in the database and critical study. Reference values have been developed by other groups within the agency, such as those found in Health Effects Assessment Documents. Most of these have not been verified or assigned an overall confidence, but in some cases (eg. cadmium)

these are the only reference values currently available.

Most RfDs developed by EPA are for chronic exposure via oral ingestion. Chronic oral RfD's are available for most Silver Mountain Mine contaminants. EPA has recently started to develop RfDs for inhalation, subchronic and acute exposures. Subchronic reference values are available for antimony, barium, beryllium, cyanide (free), manganese, nickel, selenium, thallium, vanadium and zinc. Except for selenium, subchronic oral RfDs were equal to or higher than chronic oral RfDs. In the case of selenium the chronic RfD (3×10^{-3} mg/kg/day) is slightly higher than the subchronic oral RfD (1×10^{-3} mg/kg/day) (USEPA, 1989b). Since these values are the same or lower (more conservative) than subchronic RfDs with the above exception, and since long term exposure could occur, noncarcinogenic effects have only been evaluated using chronic oral RfDs. As mentioned previously, nitrate, nitrite and copper have been evaluated separately since their critical effects result mainly from acute or subchronic exposure (see Appendix H and I).

Table 6.1 lists chronic oral RfD's for Silver Mountain Mine contaminants. For each contaminant the RfD, critical effects, uncertainty or modifying factors, and level of confidence are listed if available.

EPA verified RfDs in the Integrated Risk Information System (IRIS) database are used in the risk assessment in preference to other RfDs. For chromium, cyanide, mercury and thallium, RfDs are available for more than one chemical species. Since chemical analysis for Silver Mountain Mine did not include speciation, it has been assumed that the most toxic form of each element could be present, and therefore the lowest RfD was used.

6.2.2 Carcinogenic Effects

Since for most chemicals it is not possible to determine if there is a threshold of carcinogenesis, EPA policy is to assume that no threshold exists, although this is the subject of much debate. In other words even at low doses there is some risk of developing cancer, although the risk may be quite small. As a result, developing an RfD based on an assumed "no effects" threshold is not appropriate for evaluating carcinogenic effects.

EPA uses two steps to evaluate carcinogenic risk. First, it is determined how likely it is that the chemical will cause cancer in humans. Each chemical is given a "weight of evidence" classification depending on how strong the human, animal or other evidence is that the chemical will cause cancer. The weight of evidence classification scheme is listed below (USEPA, 1986).

The second step is to quantitatively define the dose response relationship for that chemical using one of several extrapolation models. As a matter of policy, the linearized multistage model is most often used, in part due to its conservative nature. Other models have been used, and some are still under consideration, e.g., the two stage model. The values derived from these models, or slope factors $(\text{mg/kg-day})^{-1}$, are usually only calculated for chemicals which are known or probable human carcinogens (group A, B1, or B2) and in some cases for possible human carcinogens (group C). Since there is considerable uncertainty in the modeling methods used to derive slope factors, the upper 95% confidence limit on these slope factors is used to estimate human risk. EPA believes that risk estimates derived using slope factors from the linearized multistage model are upperbound estimates. The true risk is not likely to exceed this estimate, and may in fact be zero.

For some chemicals, evidence that the chemical causes cancer has only been found for one route of exposure, e.g., via inhalation. Five chemicals found at Silver Mountain Mine have been found to be carcinogenic. Table 6.3 lists their weight of evidence classification, potency, and route by which they are believed to be carcinogenic. As described in Section 6.2.4.1, air emissions of contaminants from the site are not expected, therefore carcinogenic effects via inhalation have not been evaluated.

EPA WEIGHT OF EVIDENCE
CLASSIFICATION SYSTEM FOR
CARCINOGENICITY

<u>Group</u>	<u>Description</u>
A	Human carcinogen
B1 or B2	Probable human carcinogen
	B1 indicates that limited human data are available
	B2 indicates sufficient evidence in animals and inadequate or no evidence in humans
C	Possible human carcinogen
D	Not classifiable as to human carcinogenicity
E	Evidence of noncarcinogenicity for humans

Arsenic and lead have been found to be carcinogenic via oral ingestion. Carcinogenic risks from consuming arsenic contaminated water and soil have been evaluated using the potency listed in Table 6.3. A slope factor for lead is not available at this time to estimate carcinogenic risk, so only arsenic risks are quantitatively evaluated.

6.2.3 Chemicals of Special Concern

6.2.3.1 Arsenic

Currently there is no chronic oral Rfd for arsenic. The previously verified Rfd (1×10^{-3} mg/kg/day) has been withdrawn. This value, which is supported by the Agency for Toxic Substances and Disease Registry (ATSDR) Toxicological Profile for Arsenic, is used to evaluate the noncarcinogenic effects of arsenic since no alternative value exists. EPA is in the process of reviewing data regarding arsenic toxicity. Currently, background information pertaining to the Taiwanese epidemiologic study used to establish the previous Rfd is being reanalyzed (USEPA, 1989h).

Carcinogenic risk evaluation for arsenic differs somewhat from most other chemicals. Arsenic exposure results primarily in an increased incidence of skin cancers, only a fraction of which are fatal (USEPA, 1987). Most other forms of cancer are considered to be fatal. Arsenic also causes cancer in internal organs, but this dose response relationship is not quantifiable, and the slope factor is based solely on the incidence of skin cancers. Therefore, arsenic cancer risk estimates presented later reflect only the additional risk of skin cancer.

6.2.3.2 Lead

While a great deal of information exists regarding the toxic effects of lead, at the present time there is neither a verified Rfd nor a slope factor. EPA is attempting to develop an Rfd for lead, but for some biochemical changes (noncarcinogenic) it appears that there is no threshold below which effects do not occur (USEPA, 1989f).

The EPA Carcinogen Assessment Group has reviewed studies regarding the carcinogenesis of lead. Their review indicates that our current methods of estimating cancer risk may not be appropriate for lead, and they recommend that numerical estimates not be used.

To characterize the risk of lead exposure in water (Section 6.4), a range of drinking water concentrations is used. On May 22, 1988, proposed revisions to the drinking water Maximum

Table 6.3. Carcinogen classification and potency.

Compound	EPA Carcinogen Classification	Exposure Route	Potency (mg/kg-day) ⁻¹	Data Source
Arsenic	A	Inhalation	5.0E+01	IRIS; 7/89
	A	Oral	1.8E+00 (5.0E-05/ug/L)	IRIS; 7/89
Beryllium	B2	Inhalation	8.4E+00	IRIS; 7/89
Cadmium	B1	Inhalation	6.1E+00	IRIS; 7/89
Chromium (VI)	A	Inhalation	4.1E+01	IRIS; 7/89
Lead	B2	Oral	Not available	IRIS; 7/89
Nickel				
soluble salts	Not evaluated	NA	NA	IRIS; 7/89
nickel carbonyl	B2	Inhalation	Not available	IRIS; 7/89
nickel refinery dust	A	Inhalation	8.4E-01	IRIS; 7/89
nickel subsulfide	A	Inhalation	1.7E+01	IRIS; 7/89

Contaminant Level (MCL) were announced (USEPA, 1988a). These would change the current MCL of 50 ug/L to 5 ug/L in source water, and set action levels of 10 and 20 ug/L in the distribution system. This rule is not finalized, but personal communication with the Office of Drinking Water (USEPA, 1989i) indicate it is likely that a single MCL in the range of 10 to 20 ug/L will be established for both source water and the distribution system. The intent of drinking water maximum contaminant levels is to prevent any known adverse health effects from drinking water exposure to the contaminant of concern. However, in establishing these levels, economic and technical feasibility issues must also be considered, for example the cost and effectiveness of treatment, and the analytical detection limits. As a result, the final level chosen as the maximum contaminant level may not necessarily be the same level desired to prevent any known adverse health effects, and should not be construed as "safe". However, in the absence of a lead reference value, potential drinking water maximum contaminant levels are presented for comparative purposes.

Due to similar difficulties in evaluating the risk of ingesting lead in soil, on September 7, 1989 EPA issued a statement regarding soil lead clean up levels at Superfund sites (USEPA, 1989j). The memo indicates that for sites at which residential use is expected, 500 mg/kg should be used as the soil clean up level. For sites where industrial use is expected, 1000 mg/kg should be used. Again, while these are not reference

values, they will be used for comparison in Section 6.4.

6.2.3.3 Nitrate/Nitrite

These compounds are known to cause methemoglobinemia, as described in detail in Appendix L. They are unique in that they effect only a narrow subpopulation; newborn infants. Infants are most susceptible to hemoglobin conversion to methemoglobin because they have higher gastrointestinal concentrations of nitrate reducing bacteria, lower enzymatic capacity to reduce methemoglobin to hemoglobin, and the presence of hemoglobin F. Craun, et. al. (1981) found that children aged 1 - 8 years drinking water containing from 22-111 mg/L $\text{NO}_3(\text{N})$ did not have a significantly higher concentration of methemoglobin than children drinking water less than 10 mg/L. Based on the above information, nitrate and nitrite risks are only evaluated for the sensitive subpopulation, infants, and a six month exposure period is assumed.

6.2.3.4 Copper

The principal adverse effects of copper are gastrointestinal disorders resulting from ingestion of high concentrations in drinking water over a short period of time (USEPA, 1985). Since there is no indication that these effects occur at lower concentrations over long exposure periods, a six month exposure period has been assumed. A shorter time period, e.g., one day, could also be used, although it will result in the same average daily dose as a six month exposure period.

Copper risks have only been evaluated via the ingestion pathway, since the effects observed appear to be directly related to the oral route of administration.

6.2.4 Dermal Contact

To evaluate carcinogenic and noncarcinogenic risks from dermal contact it was necessary to utilize chronic oral RfDs since dermal RfDs have not been developed. In most cases, oral RfDs are based on the concentration administered to the test animal, as opposed to the amount absorbed into the body. To evaluate dermal contact risks, equation 6.3 is used to calculate an absorbed dose. An adjustment must therefore be made so that the oral Rfd is expressed in terms of an absorbed dose.

For most metals, there is little good information to determine the rate at which chemicals are absorbed in the gastrointestinal tract. In the absence of chemical specific absorption rates, it has been conservatively assumed that 5% of the dose administered in the animal bioassay used to develop the

Rfd was absorbed from the GI tract (USEPA, 1989p; p. A-3).

To estimate noncarcinogenic risks from dermal contact, the oral Rfds have been multiplied by 0.05 to express them in terms of an absorbed dose using the above assumption. To estimate carcinogenic risks from dermal contact with arsenic, the oral slope factor, derived from a unit risk of $5 \times 10^{-5}/\text{ug/L}$, has been divided by 0.05 based on the above assumption.

6.2.5 Uncertainty

There are two major areas of uncertainty in the toxicity assessment in addition to those discussed in Section 6.2. The first is the lack of toxicity reference values for lead, molybdenum and cobalt. As stated previously, much is known about the toxicity of lead. In combination with assumptions regarding exposure, this has been used to develop guidance regarding acceptable levels of lead in water (MCL's) and soil (see USEPA, 1989j). Although these criteria are based in part on policy decisions and not entirely on scientific information, they have been developed to be protective of human health, and may be compared to site concentrations (Section 6.4). As a result, the lack of reference values for lead is not expected to greatly affect the uncertainty of the toxicity assessment.

Soil cleanup levels or drinking water criteria (maximum contaminant levels, etc.) have not been developed for cobalt and molybdenum as they have for lead. Maximum onsite and background concentrations for water and soil are listed below. Background soil concentrations are based on two surface (1") samples collected 500 feet north and south of the heap. Background water concentrations are based on Round 3 irrigation and residential well sampling:

	<u>Onsite</u>		<u>Background</u>	
	<u>Water</u> <u>(ug/L)</u>	<u>Soil</u> <u>(mg/kg)</u>	<u>Water</u> <u>(ug/L)</u>	<u>Soil</u> <u>(mg/kg)</u>
Cobalt	7.5	17.2	6.9 - 8.0	ND
Molybdenum	20.5	9.6	6.9 - 8.0	12

Onsite concentrations of these elements do not appear to be significantly elevated above background concentrations, though background sampling is quite limited. Using worst case exposure assumptions and maximum measured site concentrations, intake of cobalt and molybdenum is estimated to be 27 ug and 42 ug daily. This compares to the estimated average daily intake from food of

300 and 350 ug/day respectively (USEPA, 1989e; 1989g). Estimated intake of these compounds from the Silver Mountain Mine site is low compared to the average daily intake from food, which is not associated with adverse effects. Thus, the lack of toxicity information for these elements is expected to have a minor impact on the findings of the risk assessment.

The second major area of uncertainty is evaluation of risks from dermal contact. Dermal Rfd's have not been developed, therefore oral Rfd's were used by converting the orally administered dose to an absorbed dose. Since there is little data in the studies used to develop Rfd's regarding the amount of chemical absorbed, it was assumed that only 5% of the orally administered dose was absorbed in the GI tract. This assumption is believed to be conservative and could have a significant impact on the results of the risk assessment.

6.3 EXPOSURE ASSESSMENT

6.3.1 Potentially Exposed Populations

Current. The 1987 community relations plan (Woodward-Clyde Consultants, 1987) and documentation for the NPL listing of the site provide information about the current population and demography in the site vicinity. No significant changes in the population distribution of the area are believed to have occurred since the information was assembled.

Within a three-mile radius of the site, fewer than 20 people are served by water supply wells. The land immediately surrounding the site is owned by a Loomis resident who uses the land for cattle grazing. The nearest residence is a single family dwelling on a farm three miles south of the site. At this location a domestic well (sampled during the Remedial Investigation) serves the residence, and a larger well supplies water for irrigation. The nearest well, used for cattle watering and for irrigation, is approximately two miles from the site. The site is located midway between Loomis (population 200) and Tonasket (population 1055). The largest town in Okanogan County is Omak (population 4,000), 26 miles south of the site.

Use of the site by local teenagers has been reported by the land owner. Early reports indicate that warning signs posted around the site were removed more than once. Ecology records also document that after the placement of the pond and heap cover, much of the rope used to hold this down was removed. Based on the above information, only infrequent visitors to the site are thought to be currently exposed.

Future. It is expected that the site will continue to be accessible to visitors in the future assuming there is no cleanup or remedial action. Others who could be exposed in the future include workers at the site or residents if people choose to live there. If it becomes profitable to continue the original mining activity, exposure to workers, ie. miners, would be a distinct possibility. Workers and residents are expected to spend far more time at the site than infrequent visitors, and as a result will be at greater risk. Since current exposures are low in comparison to potential future exposures, only future exposure scenarios will be quantified in the risk assessment.

6.3.2 Reasonable Maximum Exposure (RME)

Draft revisions of the preamble to the National Oil and Hazardous Substances Pollution Contingency Plan (NCP) (USEPA, 1989L) indicate that remedial actions at Superfund sites should be based upon the "reasonable maximum exposure". This is the highest exposure reasonably expected to occur at the site, and is intended to protect currently exposed individuals as well as those who may be exposed in the future. The method of establishing the "reasonable maximum exposure" for Silver Mountain Mine follows.

Of the current and future potentially exposed populations described in 6.3.1, workers or residents exposed in the future are expected to be at higher risk than those who currently visit the site only occasionally. Therefore, estimating exposure and risk based on future exposure scenarios is expected to be protective of both currently and potentially exposed populations, and will be used to develop the reasonable worst case.

As far as could be determined, the Silver Mountain Mine site has not been occupied in recent times. At present the nearest residence is three miles away, roughly in the center of the Horse Springs Coulee. The nearest population center, Tonasket, is six miles distant. Residential growth into the immediate vicinity of the Silver Mountain Mine site, though possible, does not appear likely in the near future. In addition, ground water availability in the immediate vicinity of the mine is very low in comparison to the center of the coulee (Chapter 3), making the site less desirable for residential occupation than areas of greater groundwater availability, assuming groundwater is used as a drinking water supply.

The site has a history of industrial use (mining) beginning in 1902 with the Silver Star Mine, and most recently the cyanide leach operation in 1981. It is possible that mining activities could occur again if it became profitable. Given the previous history of the mine, limited ground water availability, lack of

current or previous occupation, limited current exposure, and sparse residential population nearby, an industrial (eg., mining worker) exposure scenario was chosen as the reasonable maximum exposure.

The Human Health Evaluation Manual (USEPA, 1989p) suggests using all upperbound exposure parameters except for body weight, skin surface area, media concentration, and in some cases exposure duration, to develop the reasonable maximum exposure. For the Silver Mountain Mine site, average body weight and skin surface area, and upperbound worker exposure duration (40 years) were used. The upper 95% confidence limit on the arithmetic average was used for media concentration, unless it exceeded the maximum value, in which case the maximum value was used. Upperbound values were used for all other exposure parameters.

Average and upperbound exposure conditions are also presented for both the worker and residential scenarios to provide a sense for the uncertainty in the analysis and potential range in risks. These use average and either 95th percentile or maximum values for each exposure parameter respectively. Exposure parameter values for average, upperbound and reasonable maximum exposure conditions are listed in Appendix G.

6.3.3 Exposure Pathways and Routes

A conceptual diagram showing contaminant sources, release mechanisms, transport processes, pathways and receptors is shown in Figure 6.1. Contamination is believed to originate from four main sources, the leach heap, mine dump, mine drainage, and bedrock. Leaching, weathering, erosion, infiltration and other processes and mechanisms have intermixed contamination from man-made and natural sources, and transported it to other media. Analytical results have verified contamination in ground water, surface water and soil.

There are two main routes of human exposure to water. First, contaminated water, either from the shallow aquifer underlying the heap, or originating in the mine workings (which currently feeds the stock tank) could be used as a drinking water supply. Second, bathing/showering in contaminated water would result in dermal exposure. Since there currently is no agreed upon method of evaluating dermal exposure to waterborne inorganic contaminants, the dermal pathway will not be further evaluated. Other water pathways not believed to be significant are discussed below.

In some cases if contaminants in water are volatile, they may also be inhaled during bathing, showering or from other household water uses (eg. laundry, etc.). However, with the

exception of cyanide, inorganic contaminants found at Silver Mountain Mine are not volatile under normal conditions. Cyanide in the form of HCN is volatile, however cyanide found in ground water exists primarily as stable iron complexes (section 5.3.2.1) which are not volatile. Thus, inhalation from water used for domestic purposes is not considered an important pathway.

Volatilization of cyanide from leachate is evaluated in section 6.3.5.1.2. Using short term peak air concentrations from that analysis and assuming a 70 kg individual is exposed 50 feet from the heap and he/she inhales 30 m³/day, the daily dose would be 3×10^{-4} mg/kg-day. This compares to the chronic oral Rfd for free cyanide of 2×10^{-2} mg/kg-day. Assuming exposure to either long term average concentrations, or a location 200 feet from the heap will reduce the daily dose by about an order of magnitude. Assuming the more realistic heap emission rate of 96 mg/day will further reduce the daily dose by two orders of magnitude. Since these exposures are well below the free cyanide Rfd, and conservative assumptions have been used in the screening analysis, inhalation of volatilized CN from the heap is not expected to be a significant pathway of exposure.

Livestock currently use the stock tank as a watering point. Arsenic found at elevated concentrations in the stock tank is ingested by these livestock and people could be exposed if these animals are slaughtered and consumed. Bioconcentration factors for arsenic in cattle were not located in the literature, but the Canadian Government has published water quality guidelines for livestock watering (Canadian Council of Environment and Resource Ministers, 1987) which are intended to provide protection for livestock and the consumer. For arsenic, the guidelines are 500 ug/L if there are other sources of arsenic intake, or 5000 ug/L if there are few other sources. The maximum concentration of arsenic in the stock tank is 95 ug/L, well below either of the Canadian Guidelines. Therefore, consumption of locally grown cattle which drink this water is not considered a significant exposure pathway.

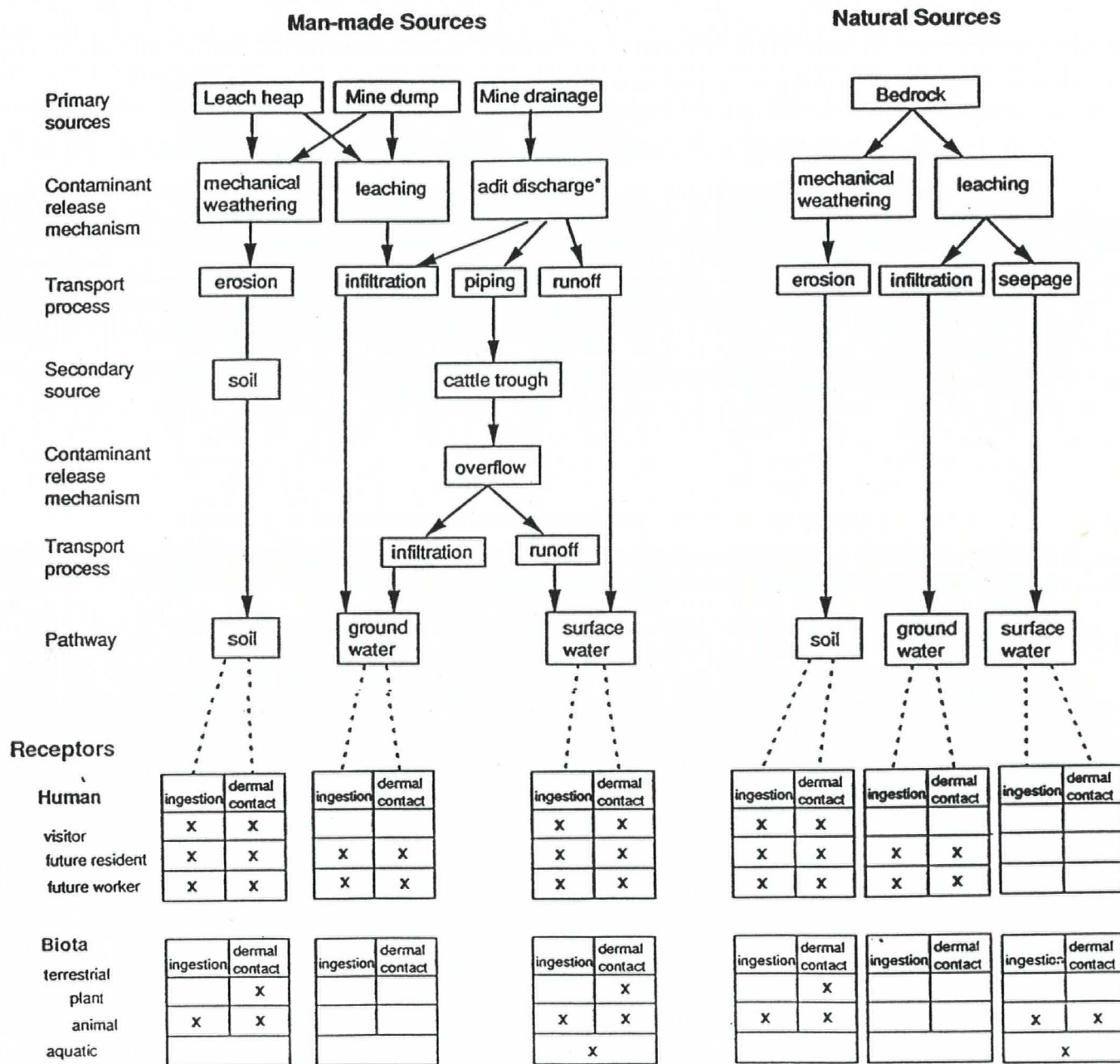


Figure 6.1 Conceptual Pathway Model.

Potential exposure pathways for contaminants in soil (eg. in the mine dump, in the heap, and near the heap) include inadvertent ingestion, (eg. while eating, smoking, etc.) direct dermal contact, or inhalation of suspended particulates (soil). As discussed in section 6.3.5.1, it is unlikely that soil particulates will be inhaled unless the heap is disturbed. The heap and mine dump have not been disturbed since the heap leach operation (other than for soil sampling by the U.S. Bureau of Mines) and there are no known plans to move or otherwise disturb the pile, therefore it has been assumed for the risk assessment that the pile will not be disturbed and there will be no remedial action. Therefore the main soil exposure pathways are ingestion and dermal contact.

6.3.4 Exposure Point Concentrations

Water and soils sampling data collected during 1988 and 1989 were used to estimate chemical concentrations. These data are summarized below.

6.3.4.1 Water

Three rounds of water sampling were conducted as described in Chapter 4. In all three rounds metals and anions were analyzed from the four onsite monitoring wells, nearby stock watering tank, nearby mine seep, and from offsite irrigation wells and the nearest residence. Tables 4.2 through 4.6 list these data. Due to limitations in the quality of some of the data, only a subset could be used for the quantitative risk assessment. Data with an assigned "R" qualifier have not been used. Data with an assigned "J" qualifier have been used, although these concentrations are only estimated.

Data from the four on-site monitoring wells were used to evaluate the risk of consuming groundwater from the immediate vicinity of the leach heap and mine dump. Risk from consuming water in the stock tank was also evaluated. Only Round 3 metals results were used, as Round 1 samples were not field acidified and acid used to preserve Round 2 samples was contaminated. All three rounds of anion data were usable for fluoride, and Round 2 and 3 data were usable for cyanide. Nitrate was only analyzed in Rounds 2 and 3 (both usable), and nitrite only in the last round.

To estimate average exposure conditions, arithmetic average concentrations were calculated for the data just described. To estimate worst case (upperbound) exposure conditions 95th percentile concentrations are used (USEPA, 1989a). If 20 data points are available, then each is equivalent to a 5th percentile category and the 95th percentile may be determined. With fewer than 20 data points, the 95th percentile value is between the two

highest values. Since the distribution of values between these points is unknown, there is uncertainty in estimating (interpolating) the 95th percentile, which increases as the number of samples decreases. Since at most 12 data points were available, the maximum value from the usable data was used to estimate upperbound exposure conditions, rather than estimating the 95th percentile. Table 6.2 lists average and upperbound water concentrations for each chemical evaluated quantitatively.

6.3.4.2 Soil

As described in Chapter 4, one round of soils samples were collected from the heap, beneath the trench south of the heap, the mine dump, and from nearby soil. Samples from the mine dump and heap were collected from a variety of depths, from two feet to six feet below the surface. Only the soil fraction of these samples were analyzed. The top one inch of soil from sites near the heap and mine dump, and background samples 500 feet north and south of the heap were also collected. As with water data, values with an assigned "R" qualifier have not been used in the quantitative assessment, whereas unqualified data and data with a "J" qualifier have been used.

Data from all sampling locations, except background, have been grouped together to estimate potential future exposures via the soil pathway. The arithmetic average concentration was used to estimate average exposure, and the 95th percentile concentration was used to estimate the upperbound exposure. These values are listed in Table 6.2.

6.3.5 Contaminant Fate/Transport

Analytical data for soil and water collected during 1988 and 1989 represent current contaminant concentrations in these media. To fully characterize current and future levels of contaminants, in all media, information regarding contaminant concentrations in air, and the fate and transport of contaminants in all media are necessary. Since air monitoring was not conducted at this site, it was necessary to model air concentrations.

6.3.5.1 Air Fate/Transport

6.3.5.1.1 Particulate emissions

To determine potential particulate emissions from the heap under worst case conditions, a screening analysis was performed. Appendix M is a detailed discussion of the screening analysis. Using site information and conservative assumptions it was determined that particulate emissions from the heap are unlikely. In the analysis the assumption that the heap is not disturbed and

Table 6.2. Contaminant concentrations used to estimate risk.

CONTAMINANT	SOIL (mg/kg)			WATER (ug/l)		
	AVE.	U.B.	RME	AVE.	U.B.	RME
Antimony	3.0	9.1	4.9	14.7	40.4	40.4
Arsenic	342.5	631.6	420.8	10.7	14.3	14.3
Barium	53.2	109.5	60.9	61.0	136.0	136.0
Beryllium	0.3	0.7	0.4	0.9	1.5	1.5
Cadmium	2.0	4.2	2.4	1.5	2.9	2.9
Chromium	10.4	16.0	11.8	11.1	31.6	31.6
Copper	134.7	510.4	185.6	20.2	56.7	56.7
Cyanide	21.9	96.3	35.1	40.8	281.0	122.3
Fluoride *	0.0	0.0	0.0	0.4	0.7	0.5
Lead	82.4	193.9	103.5	8.7	23.2	23.2
Manganese	576.7	938.0	630.9	166.0	421.0	421.0
Mercury	0.2	0.6	0.3	0.1	0.1	0.1
Nickel	29.6	48.8	33.3	15.6	38.4	38.4
Nitrate *	0.0	0.0	0.0	37.1	120.8	76.3
Nitrite *	0.0	0.0	0.0	0.3	1.3	1.3
Selenium	0.8	1.6	1.0	1.7	3.9	3.9
Silver	8.4	33.8	11.9	2.5	2.5	2.5
Thallium	0.2	0.9	0.3	0.1	0.1	0.1
Tin	0.0	0.0	0.0	15.4	31.5	31.5
Vanadium	18.8	30.6	21.8	11.5	40.7	40.7
Zinc	224.3	554.1	274.9	42.1	129.0	129.0

* Water concentrations are in mg/l.

RME = Reasonable maximum exposure

Ave. = Average

U.B. = Upperbound

the surface characteristics of the heap (roughness) were significant factors preventing emissions. Since heap materials originated from the mine dump, surface characteristics of the mine dump are expected to be very similar to materials in the heap. Therefore, particulates are not expected to be suspended from the mine dump.

Although an emissions analysis of nearby soil was not carried out, due to the limited extent of contamination beyond the heap and mine dump, particulate emissions from the nearby soil are not expected to be significant.

In summary, suspension of soil from the heap, mine dump, and nearby soil is not expected to be a significant transport mechanism.

6.3.5.1.2 Volatilization of cyanide

From Section 4.4.1.1, cyanide in the heap soils (solids) exists mainly in stable iron complexes. These may be degraded by sunlight to form, in part, HCN which readily volatilizes (USEPA, 1979). The heap has been exposed to sunlight for 3 years, from 1981 to 1984, and has been chemically treated twice with sodium hypochlorite (HTH) to destroy cyanide. Therefore, little if any HCN is expected to be produced by photodecomposition at the heap's surface in the future. Only cyanide forms dissolved in leachate migrating from the heap are expected to be available to be converted to HCN.

Several conservative assumptions have been made to evaluate the significance of this occurring. First, from section 5.3.2.2, it was assumed that cyanide concentration in leachate is the maximum probable, 2.2 mg/L. Second, it was assumed that seepage velocity from the leach heap was the maximum probable, 0.3 m/day (section 5.2.1), giving 9600 liters/day when evaluated over the area of the heap base (3200 m²). From Figure 5.2, a daily degradation rate of 0.01 mg/L CN per day was determined assuming the leachate concentration of 2.2 mg/L. It was assumed that all of the cyanide lost was converted to HCN, all of which volatilized to the atmosphere. Given these assumptions, it was estimated that 9,600 mg/day of CN would volatilize from the heap. Assuming a more realistic infiltration rate of 0.03 m/day, 96 mg/day of CN would be emitted. The maximum worst case value of 9600 mg/day was used in the SCREEN dispersion model (USEPA, 1988e) to provide a worst case estimate of air concentrations over the nearby area, further assuming a worker or resident was 50 or 200 feet downwind of the heap. Results from this analysis follow:

Distance downwind (feet)	Short term (1 hour) (ug/m ³)	Long term (15% of 1 hour) (ug/m ³)
50	0.53	0.079
200	0.206	0.031

These results indicate that only very low concentrations of HCN would be emitted from the heap using worst case assumptions.

6.3.5.2 Water Fate/Transport

The fate and transport of contaminants in water, focusing mainly on arsenic and cyanide, are described in detail in Chapter 5. Projected future cyanide concentrations in ground water onsite are expected to be lower than the current maximum concentration of 0.28 mg/L. As ground water moves offsite into the Horse Springs Coulee aquifer, concentrations will further drop.

Arsenic concentrations may increase with time. Concentrations as high as 27 mg/L may recharge the shallow aquifer onsite, and concentrations as high as 0.03 mg/L may seep into the Horse Springs Coulee aquifer.

6.3.5.3 Soil Fate/Transport

The principal fate and transport mechanisms of soil contaminants are wind erosion and leaching from precipitation. Suspension of soil particles in air by wind erosion is discussed in section 6.3.5.1.1, and leaching of soil contaminants is described in Chapter 5 and section 6.3.5.2.

6.3.6 Estimate Chemical Intake

Standard exposure equations found in current (USEPA 1989a; 1989p) and previous EPA risk assessment guidance are used to estimate chemical intake by water ingestion, soil ingestion and dermal contact with soil. These equations are used to calculate an average daily intake over the exposure period of interest in terms of milligrams (mg) of contaminant per kilogram (kg) of an individual's body weight per day (mg/kg/day). Pathway specific exposure equations are as follows:

Ingestion of chemicals in drinking water

$$\text{Eq. 6-1} \quad \text{Intake (mg/kg-day)} = \frac{\text{CW} \times \text{IR} \times \text{EF} \times \text{ED} \times \text{CF}}{\text{BW} \times \text{AT}}$$

Where:

CW = chemical concentration in water (ug/liter)
IR = ingestion rate (liters/day)
EF = exposure frequency (unitless)
ED = exposure duration (days)
CF = conversion factor (10^{-3} mg/ug)
BW = body weight (kg)
AT = averaging time (days)

Ingestion of chemicals in soils

$$\text{Eq. 6.2} \quad \text{Intake (mg/kg-day)} = \frac{\text{CS} \times \text{IR} \times \text{CF} \times \text{FI} \times \text{EF} \times \text{ED}}{\text{BW} \times \text{AT}}$$

Where:

CS = chemical concentration in soil (mg/kg)
IR = ingestion rate (mg soil/day)
CF = conversion factor (10^{-6} kg/mg)
FI = fraction ingested from contaminated source
EF = exposure frequency (unitless)
ED = exposure duration (days)
BW = body weight (kg)
AT = averaging time (days)

Dermal contact with soils

$$\text{Eq. 6.3} \quad \text{Intake (mg/kg-day)} = \frac{\text{CS} \times \text{CF} \times \text{SA} \times \text{AF} \times \text{ABS} \times \text{EF} \times \text{ED}}{\text{BW} \times \text{AT}}$$

Where:

CS = chemical concentration in soil (mg/kg)
CF = conversion factor (10^{-6} kg/mg)
SA = surface area available for contact (cm^2)
AF = soil to skin adherence factor ($\text{mg}/\text{cm}^2/\text{day}$)
ABS = absorption factor through skin (unitless)
EF = exposure frequency (unitless)
ED = exposure duration (days)
BW = body weight (kg)
AT = averaging time (days)

Exposure parameter values (eg., water ingestion rate, body surface area, etc.) previously established by EPA Region 10 for average, upperbound, and reasonable maximum exposure conditions were used for this assessment. These values, and the rationale and references used in their derivation, are found in Appendix G. Where appropriate data existed, 50th percentile and 95th percentile values were used to estimate average and upperbound exposures, respectively. A combination of upperbound and average exposure parameters were used to develop the maximum exposure, as described in Section 6.3.2

Doses were calculated separately to estimate noncarcinogenic and cancer risks using two models developed by EPA Region 10 (USEPA 1989o; 1989m). Using the first model, dose is estimated for noncarcinogenic endpoints by averaging chemical intake over the critical period of exposure for each pathway. For chemicals other than copper, nitrate and nitrite, long term (chronic) exposure is the most critical, and chemical intake was averaged over 7 years. The dose received (mg/kg/day) from water or soil ingestion is highest for children since their ingestion rate is high relative to their body weight when compared to adults. Averaging the dose received during childhood therefore provides an estimate of dose for this sensitive subpopulation. Averaging dose received during childhood and adulthood, or adulthood alone, gives a less conservative estimate of the dose received. Since seven years is the shortest exposure period to which it is appropriate to apply a chronic oral Rfd (USEPA, 1989p), noncarcinogenic effects are evaluated by averaging the dose received by a child exposed from age 0-7 years.

Adverse effects of copper, nitrate, and nitrite result principally from short term (acute or subchronic) exposure, therefore average intakes were calculated separately for these chemicals using a 6 month averaging period. See Appendix H and I for dose and risk calculations for these chemicals.

The second model is used to estimate dose for each pathway for cancer risk estimation. Current EPA methodology is to average chemical exposure over an individual's lifetime to derive this dose, regardless of the exposure duration or concentration (USEPA, 1989p). A short term exposure to high concentrations is believed roughly equivalent to long term exposure to low concentrations. In practice, dose is summed over the exposure period, e.g., 9 years for average residential exposure, and then is averaged over a person's life (75 years). In contrast, dose would only be averaged over the given exposure period, 9 years in this case, using the noncarcinogenic model.

These models are used to estimate doses for each chemical by pathway and scenario, as well as providing risk estimates, discussed further in section 6.4. Appendices J and K are

spreadsheets from the noncancer and cancer models (respectively) for arsenic, which show both dose and risk estimates. Doses for other chemicals except copper, nitrate and nitrite were calculated in the same manner.

To determine the overall chemical intake, doses must be summed across all relevant pathways. In this case it is reasonable to assume that a person will be concurrently exposed to contaminants by three routes; soil ingestion, dermal contact with soil, and ingestion of drinking water. Therefore, doses from these pathways have been added to determine the cumulative dose.

6.4 RISK CHARACTERIZATION

6.4.1 Definition of Risk Endpoints

6.4.1.1 Carcinogenic Effects

Risk is calculated and presented differently for carcinogenic and noncarcinogenic endpoints. The average dose, as described in section 6.3.6, is used to estimate exposure in both cases. Carcinogenic risk is usually calculated and presented as the increased probability that a person will contract cancer as a result of exposure to a chemical, (ie. individual risk). As described in section 6.2, current EPA policy is that exposure to carcinogens, at any level, presents some risk. The formula to estimate this risk is:

$$\text{Risk} = \text{ADI} \times \text{SF}$$

where:

Risk = a unitless probability (eg. 2×10^{-5}) of an individual developing cancer;

ADI = average daily intake averaged over a persons life (75 years) in mg/kg-day

SF = slope factor (mg/kg-day)⁻¹.

From section 6.2.2, the upper 95% confidence limit of results from the linearized multistage model are most often used by EPA to derive chemical-specific slope factors. As a result, when these slope factors are combined with average daily intakes the risk estimates calculated are believed to be upperbound values. EPA believes that it is unlikely that the true risk will exceed this risk estimate, and may in fact be zero.

Another way to present cancer risk estimates is in terms of additional cancer cases that may occur as a result of exposure to a chemical, ie. population risk. It is most meaningful to present risk in these terms when a large population is being (or is potentially) exposed. Few people are currently exposed to contamination at Silver Mountain Mine and it is not expected that this will significantly change in the future. Therefore, only individual risk is presented.

6.4.1.2 Noncarcinogenic Effects

Risk of developing noncarcinogenic effects are presented in terms of a ratio of the average dose received over the exposure period, divided by the Rfd for the appropriate exposure period. This ratio is known as the noncancer hazard quotient and is calculated as follows (USEPA, 1989p):

$$\text{Noncancer hazard quotient} = E/Rfd$$

where:

E = exposure level (mg/kg-day)

Rfd = reference dose (mg/kg-day)

As discussed previously, it is believed that there is a threshold of exposure to chemicals below which it is not expected that there will be adverse effects. Above this threshold, noncancerous effects may occur. The Rfd is intended to represent the threshold of exposure below which adverse effects will not occur. If the exposure is equal to or less than the Rfd, ie. a hazard quotient of 1.0 or less, then adverse effects are not expected. If exposure exceeds the Rfd, then there is an increasing chance that adverse effects will occur. The probability of these effects is not calculated, but several factors affect the interpretation of hazard quotients greater than 1.0.

First, the slope of the dose response curve at doses above the reference dose will determine how likely adverse effects will be. These slopes will vary from chemical to chemical, suggesting that hazard quotients greater than 1.0 are not directly comparable between chemicals.

Second, depending on the strength of data used to establish the Rfd, different uncertainty factors have been used. If good human data are used, the combination of uncertainty factors may be equal to 1. If poor data are used, it may be 1,000 or 10,000. Therefore, while it is desirable not to exceed a hazard quotient of 1, the consequences of doing so if an Rfd has an uncertainty factor of 1 could be more serious than doing so for a

chemical with an uncertainty factor of 1,000.

Finally, the level of confidence in the database used to establish the Rfd will affect interpretation of hazard quotients. Clearly if there is low confidence in the database, exceeding a hazard quotient of 1 may be less serious than if there is high confidence in the database. EPA identifies the level of confidence in the database when it verifies Rfd's. When available, the level of confidence has been listed in Table 6.1 .

Noncarcinogenic risk from exposure to multiple chemicals is evaluated by adding their respective hazard quotients, to derive a hazard index as follows:

$$\text{Hazard Index} = \text{HQ}_1 + \text{HQ}_2 + \dots \text{HQ}_i$$

where:

$\text{HQ}_{1..i}$ = the hazard quotient for 1st through i^{th} chemical

These are interpreted similarly to the hazard quotient. First it is assumed that risks from exposure to multiple chemicals are additive, rather than independent, synergistic, or antagonistic. If the hazard index does not exceed 1, it is not expected that there will be adverse effects. As the hazard quotient increases above 1, noncancer effects become more likely. Generally it is desirable to have a hazard index of 1 or less.

It is most appropriate to add risks across chemicals using the hazard index if the chemicals affect the same target organ, and have similar mechanisms of action (USEPA, 1989p). The procedure used to do so with Silver Mountain Mine contaminants is described further in section 6.4.3.4.

6.4.2 Carcinogenic Risk

Of the contaminants identified at Silver Mountain Mine, arsenic, beryllium, cadmium, chromium, lead, and nickel are known or probable human carcinogens. Beryllium, cadmium, chromium, and nickel are only carcinogenic via inhalation, which is not a significant pathway (section 6.3.3), and have not been evaluated. Lead could not be evaluated due to the lack of a slope factor. The following discussion is based entirely on risk estimates calculated for arsenic.

6.4.2.1 Risk by pathway

Cancer risk estimates for both the industrial scenario and residential scenario, both average and upper bound cases, and for the reasonable maximum exposure, are presented in Table 6.4. Total risk as well as risk by pathway is presented.

Table 6.4. Arsenic carcinogenic risk.

PATHWAY	RESIDENTIAL		INDUSTRIAL		RME
	AVE.	U.B.	AVE.	U.B.	
SOIL INGESTION	7.29E-04	5.74E-03	2.94E-05	3.46E-04	2.3E-04
WATER INGESTION	7.05E-05	9.10E-04	2.05E-05	2.35E-04	2.4E-04
DERMAL CONTACT	4.53E-05	1.48E-02	8.45E-06	5.80E-03	1.9E-03
PARTICULATE INHALATION	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.0E+00
VAPOR INHALATION	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.0E+00
TOTAL CARCINOGENIC RISK	8.45E-04	1.48E-02	8.45E-06	6.38E-03	2.3E-03

Ave. = Average

U.B. = Upperbound

RME = Reasonable maximum exposure

For the reasonable maximum exposure, risks ranged from 2×10^{-3} for dermal contact to 2×10^{-4} for soil and water ingestion. For all scenarios and parameter ranges, the dermal contact route accounts for most of the risk. Dermal risks are about an order of magnitude higher than soil and water ingestion using upper bound and reasonable maximum exposure parameters. Using average exposure parameters, dermal contact risks are equal to or less than soil and water ingestion risks. Within each pathway, risks increase approximately an order of magnitude when using upper bound or reasonable maximum exposure parameters compared to average exposure parameters.

6.4.2.2 Risk by Medium

Recent revisions to the preamble of the final draft of the National Contingency Plan (NCP) indicate that risk goals for cleanup (e.g., 10^{-6}) are medium specific (USEPA, 1989L). To reflect these revisions, Table 6.5 lists carcinogenic risk by medium; soil and water. Exposure to arsenic in soil by ingestion and dermal contact results in a risk of 2×10^{-3} . Exposure to arsenic via drinking water results in a carcinogenic risk of 2×10^{-4} .

6.4.2.3 Combined Carcinogenic Risk

The overall risk (Table 6.4) from exposure to all media via all pathways ranged widely from 2×10^{-2} for the upper bound residential scenario to 9×10^{-6} for the average industrial exposure. Risks are generally higher by at least an order of magnitude for the residential versus the industrial scenario, and the reasonable maximum exposure is slightly less (2×10^{-3} vs 6×10^{-3}) than the upper bound industrial scenario.

6.4.2.4 Stock Tank

Since the stock tank could potentially be used as a human drinking water supply, a separate estimate of carcinogenic risk was calculated assuming reasonable maximum exposure parameters. The arsenic concentration in the trough (95 ug/L; Round 3 Metals Analysis) is higher than from any on-site monitoring wells. The excess cancer risk associated with drinking this water is 2×10^{-3} .

6.4.3 Noncarcinogenic Risk

Table 6.5 lists hazard quotients for all exposure scenarios by pathway, as well as the total hazard index by summing across exposure pathways. The overall hazard indices ranged from 1 (average industrial) to 129 (upper bound residential). For the reasonable maximum exposure, both water ingestion and dermal contact exceeded a hazard index of 1.0; 3.1 and 2.2 respectively. As with carcinogenic risk, the dermal pathway risks are higher than other pathways using upper bound exposure parameters, but less by an order of magnitude using average exposure parameters. Within each pathway, risks increased by an order of magnitude (residential) or less (industrial) when comparing averages to upper bound exposure parameters.

6.4.3.1 Risk by Chemical

Table 6.7 lists hazard quotients for each chemical by exposure scenario, as well as the total hazard index. By focusing on just those chemicals with a hazard quotient of 1 or greater, important noncarcinogenic contaminants can be identified. In four of five scenario and exposure parameter groupings, antimony and arsenic exceed a hazard quotient of 1. Cyanide exceeds a hazard quotient of 1 in two of the five scenarios, and is an important contributor in the reasonable maximum exposure scenario with a hazard quotient of 0.46. Nitrate and nitrite exceed 1 in the upper bound residential scenario, but do not appear in the reasonable maximum exposure scenario, since childhood exposure is not expected in that setting. Other contaminants which exceed 1 in the upper bound residential scenario include cadmium, chromium, copper, silver and thallium.

Table 6.5. Noncarcinogenic risk by pathway.

PATHWAY	<u>Hazard Quotient</u>				
	RESIDENTIAL		INDUSTRIAL		
	<u>AVE.</u>	<u>U.B.</u>	<u>AVE.</u>	<u>U.B.</u>	<u>RME</u>
SOIL INGESTION	4.7E+00	3.7E+01	1.4E-01	4.4E-01	2.7E-01
WATER INGESTION	3.3E+00	4.0E+01	5.9E-01	3.6E+00	3.1E+00
DERMAL CONTACT	2.8E-01	5.3E+01	3.9E-02	6.9E+00	2.2E+00
PARTICULATE INHALATION	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00
VAPOR INHALATION	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00
TOTAL HAZARD INDEX	8	129	1	11	5

Ave. = Average
U.B. = Upperbound
RME = Reasonable maximum exposure

6.4.3.2 Risk by Medium

While the recent revisions to the National Contingency Plan indicate that carcinogenic risk goals are medium-specific, they do not address remedial goals for noncarcinogenic effects. In the absence of such guidance, it has been assumed that goals for noncarcinogenic endpoints are medium-specific as well.

Table 6.6 lists hazard quotients by chemical and by medium for the reasonable maximum exposure. Risks from water and soil are about equal when comparing hazard indices; 3.1 and 2.4 respectively. The most important water contaminant is antimony, followed by cyanide and others. In soil, the arsenic hazard index of 2.2 accounts for a majority of the hazard quotient of 2.2.

6.4.3.3 Stock Tank

Noncarcinogenic risks which would result from consuming water in the stock tank are presented in Table 6.8. Antimony and selenium values from Round 3 were not usable, so these risks could not be calculated. Arsenic is by far the most important of the other chemicals; its hazard quotient of 1.6 accounting for a majority of the 1.7 hazard index.

6.4.3.4 Lead

As identified in Section 6.3.3.2, reference values are not available for lead to estimate risk. Instead, water and soil concentrations have been compared to drinking water criteria, and

Table 6.6. Reasonable Maximum Exposure (RME) risks by media.

A. NONCARCINOGENIC

	Rfd Ratio	
	Water	Soil
	-----	-----
Antimony	1.7E+00	6.3E-02
Arsenic	2.5E-01	2.2E+00
Barium	4.7E-02	6.3E-03
Beryllium	5.1E-03	4.2E-04
Cadmium	9.8E-02	2.5E-02
Chromium	1.1E-01	1.2E-02
Copper	2.6E-02	1.1E-04
Cyanide	4.2E-01	3.7E-02
Fluoride	2.0E-01	0.0E+00
Manganese	3.6E-02	1.6E-02
Mercury	5.7E-03	5.0E-03
Nickel	3.3E-02	8.7E-03
Nitrate	0.0E+00	0.0E+00
Nitrite	0.0E+00	0.0E+00
Selenium	2.2E-02	1.7E-03
Silver	1.4E-02	2.1E-02
Thallium	1.1E-02	1.7E-02
Tin	9.0E-04	0.0E+00
Vanadium	7.8E-02	1.3E-02
Zinc	1.1E-02	7.1E-03
	-----	-----
Hazard Index:	3.1	2.4
	=====	=====

Combined Hazard Index: 5.5

B. CARCINOGENIC (Arsenic only)

Water	Soil
-----	-----
2.4E-04	2.1E-03
	=====
Total risk:	2.3E-03

Table 6.7. Noncarcinogenic risk by chemical.

Contaminant	Hazard Quotient				
	Residential		Industrial		
	Ave.	U.B.	Ave.	U.B.	RME
Antimony	1.3E+00	1.3E+01	3.0E-01	2.0E+00	1.8E+00
Arsenic	4.8E+00	7.9E+01	2.4E-01	6.6E+00	2.4E+00
Barium	5.3E-02	5.5E-01	1.0E-02	6.9E-02	5.3E-02
Beryllium	6.5E-03	4.8E-02	1.5E-03	6.6E-03	5.6E-03
Cadmium	1.5E-01	1.6E+00	2.6E-02	1.8E-01	1.2E-01
Chromium	9.8E-02	1.0E+00	1.9E-02	1.4E-01	1.2E-01
Copper	9.2E-02	1.4E+00	1.7E-02	3.4E-02	2.9E-02
Cyanide	3.2E-01	8.2E+00	6.7E-02	1.2E+00	4.6E-01
Fluoride	1.9E-01	1.2E+00	4.8E-02	2.0E-01	2.0E-01
Manganese	6.4E-02	7.9E-01	8.0E-03	8.4E-02	5.2E-02
Mercury	1.9E-02	2.7E-01	3.0E-03	2.6E-02	1.1E-02
Nickel	4.4E-02	5.0E-01	6.9E-03	5.8E-02	4.2E-02
Nitrate	8.2E-01	1.5E+01	0.0E+00	0.0E+00	0.0E+00
Nitrite	1.0E-01	2.2E+00	0.0E+00	0.0E+00	0.0E+00
Selenium	2.1E-02	2.0E-01	4.7E-03	2.8E-02	2.4E-02
Silver	6.3E-02	1.5E+00	8.0E-03	1.3E-01	3.5E-02
Thallium	4.4E-02	1.4E+00	5.9E-03	1.2E-01	2.8E-02
Tin	8.2E-04	5.5E-03	2.1E-04	9.0E-04	9.0E-04
Vanadium	6.8E-02	8.9E-01	1.1E-02	1.1E-01	9.0E-02
Zinc	2.1E-02	4.1E-01	2.2E-03	3.9E-02	1.8E-02
Total Hazard Index:	8	129	1	11	6

=====

Ave. = Average
U.B. = Upperbound
RME = Reasonable maximum exposure

Table 6.8. Stock tank drinking water risks (RME); reasonable maximum exposure.

Noncarcinogenic Risks		Carcinogenic Risk	
Compound	Hazard quotient	Compound	Risk
-----	-----	-----	----
Antimony	0.0E+00		
Arsenic	1.6E+00	Arsenic	1.6E-03
Barium	3.4E-03		
Beryllium	1.7E-03		
Cadmium	1.7E-02		
Chromium	3.4E-03		
Cyanide	4.1E-03		
Fluoride	7.1E-05		
Manganese	2.0E-04		
Mercury	5.7E-03		
Nickel	1.1E-02		
Selenium	0.0E+00		
Silver	1.4E-02		
Thallium	1.1E-02		
Tin	2.9E-04		
Vanadium	2.4E-03		
Zinc	5.1E-04		
-----	-----		
Hazard Index =	1.7E+00		

established soil cleanup levels, respectively listed below. Drinking water criteria include the current EPA maximum contaminant level (USEPA, 1988d) proposed sourcewater maximum contaminant level (USEPA, 1988a) and an "Action Level" (USEPA, 1989k) used internally by EPA as a limit for EPA facilities. As discussed in Section 6.2.3.2, policy considerations such as cost are evaluated in establishing these levels, so they should not be considered in the same context as an unbiased, health based reference value such as an agency verified Rfd.

In the absence of an Rfd or slope factor for lead, interim soil clean-up levels (USEPA, 1989j) were established. As with drinking water maximum contaminant levels, these levels are based in part on policy decisions, as well as health related considerations.

<u>Soil Cleanup Level (1)</u>		<u>Site Concentrations</u>
Industrial	1000 mg/kg	103.5 mg/kg (RME)
Residential	500 mg/kg	193.9 mg/kg (maximum)
<u>Water Criteria</u>		<u>Site Concentrations</u>
Current MCL (2)	50 ug/L	23.2 ug/L (RME, maximum)
Action Level (3)	20 ug/L	
Proposed MCL (4)	5 ug/L	8.7 ug/L (average)
(source water)		

(1) USEPA, 1989j
(2) USEPA, 1988d

(3) USEPA, 1989k
(4) USEPA, 1988a

For soil, both reasonable maximum exposure (RME) and maximum lead concentrations on-site are below recommended cleanup levels for both industrial and residential settings, indicating lead is not a significant problem in soil.

Recent conversations with the Office of Drinking Water (USEPA, 1989i) indicate it is most likely that a single MCL of from 10 - 20 ug/L will be set for both source water and water within distribution systems. Only the maximum concentration in monitoring well 1 (23.2 ug/L) exceeds this range, the next highest measured concentration was 6.5 ug/L.

As mentioned in Chapter 2, monitoring well 1 has very low water yield. Due to the low yield, it was difficult to properly develop, and samples from it were turbid. As a result, data from this monitoring well may not be representative of actual site conditions. Also, this well is upgradient from the leach heap,

mine and mine discharge, suggesting that lead found in it may originate in local bedrock or sediment.

6.4.3.5 Summarizing by Critical Effect

Since hazard indices for both soil and water are greater than 1.0, the hazard quotient was further refined to ensure that only chemicals with similar toxic effects are grouped together.

A matrix of critical effects and interactions was first derived by reviewing toxicological profiles in Appendix L and critical effects in Table 6.1. Compounds were grouped based on similar patterns of critical effects and interactions. These are listed in Table 6.9. In some cases, e.g., antimony and arsenic, chemicals were grouped because they are known to have similar toxic properties, although data are quite limited (antimony). Other chemicals including fluoride, silver, cobalt, tin, beryllium and nickel are thought to have toxic effects via ingestion that are not similar to other chemicals, so they are listed separately.

These groupings are based mainly on critical effect organ, with some evaluation of the mechanisms involved. Some chemicals, eg. lead, have many toxic endpoints. Rather than list all possible effects, those which appear to be most adverse and characteristic of the compound have been listed. Such a procedure may overlook important pharmacokinetic information which could suggest further splitting (or lumping) of groups. A detailed pharmacokinetics evaluation was not carried out for this assessment. Nonetheless, the hazard indices presented in Table 6.9 are believed to be more valid indicators of noncarcinogenic risk than simply adding hazard quotients for all chemicals.

The grouping of antimony, arsenic, manganese, cyanide, selenium and barium exceeded a hazard index of 1.0 in both water (2.5) and soil (2.3). None of the other groupings exceeded 1.0. This analysis supports the earlier findings that arsenic, antimony and cyanide are the most important contaminants, since arsenic and antimony have hazard quotients greater than 1.0, and all have similar toxic endpoints.

6.5 UNCERTAINTY ASSESSMENT

The uncertainty associated with this type of risk assessment is inherently large, i.e., an order of magnitude or more (USEPA, 1989p). To evaluate uncertainty in the Silver Mountain Mine assessment, the key assumptions and data used have been assigned relative rankings (low, moderate, high) regarding their potential to overestimate risk based on best professional judgement. In terms of order of magnitude impact on risk, these are;

Table 6.9. Noncarcinogenic risk grouped by critical effects.

Effects	Compounds	Water Hazard Index	Soil Hazard Index
Nerve, liver, skin	antimony, arsenic, manganese, cyanide, selenium, barium, lead	2.48	2.32
Fluorosis	fluoride	0.2	NA
Argyria	silver	0.014	0.021
Methemoglobinemia	nitrate, nitrite		NA
Kidney	cadmium, zinc, mercury, lead, molybdenum	0.11	0.037
Polycythemia, thyroid, cardiac, bone marrow	cobalt	NA	NA
Gastrointestinal	copper, vanadium	0.1	0.013
Alopecia, enzyme effects	thallium, vanadium	0.09	0.030
Immunotoxicity	tin	0.0009	NA
Skin, lung	beryllium	0.0051	0.00042
Contact dermatitis, non-specific	nickel	0.033	0.0087
=====			

Table 6.10. Uncertainty assessment.

Component	POTENTIAL TO OVER OR UNDER ESTIMATE RISK*		
	Over	Under	Unknown

Chemical Selection			
- deleting nutrients		Low	
- delete chemicals w/no tox. values		Low	
Toxicity Assessment			
- arsenic			Moderate
- antimony	High		
- cyanide	High		
- chemical interactions			Low/Moderate
Data			
- representativeness			Low
- quality			Low
Fate/Transport Modeling			
- air		Low	
- water	Low/Moderate		
Exposure Parameters			
- industrial scenario	Low/Moderate		
- residential scenario	High		
- dermal absorption	High		
- soil data	Low		
- water data	Moderate		

=====

* Values assigned based on best professional judgement.

Low = <0.5 order of magnitude

Moderate = 0.5 - 1.0 order of magnitude

High = >1.0 order of magnitude

Table 6.11. Drinking Water Regulations and Guidance.

Chemical	Groundwater Concentration (ug/L; RME)	Current MCL (1) (ug/L)	Proposed MCL (2) (ug/L)	Lifetime Health Advisory (ug/L)(3)	Other (ug/L)
Antimony	40.4				
Arsenic	14.3	50			
Barium	136	1000	5000	1500	
Beryllium	1.5	10			
Cadmium	2.9	50	5	5	
Chromium	31.6	50	100	120	
Copper	56.7		1300		
Cyanide	122.3			154	
Fluoride*	0.5	4			
Lead	23.2	50	5 - 20 (5)		20 (4)
Manganese	421				
Mercury	0.1	2	2	1.1	
Nickel	38.4			150	
Nitrate (as N)*	76.3	10	10	10	
Nitrite (as N)*	1.3		1	1	
Selenium	3.9	10			
Silver	2.5	50	Delete		
Thallium	0.1				
Tin	31.5				
Vanadium	40.7				
Zinc	129				

* - units are mg/L

MCL = Maximum contaminant level

RME = Reasonable maximum exposure

- (1) USEPA. National Primary Drinking Water Regulations. 40 CFR Part 141.
- (2) USEPA. National Primary and Secondary Drinking Water Regulations; Proposed Rule. Federal Register Vol. 54, No. 97. May 22, 1988.
- (3) USEPA. Final Health Advisories. Office of Drinking Water. March 31, 1987.
- (4) USEPA. April 24, 1989 Memorandum. From: John C. Chamberlin, Director, Office of Administration and Michael B. Cook, Director, Office of Drinking Water, To: Regional Administrators, et al.; Subject: Monitoring Drinking Water for Lead at EPA Facilities.
- (5) USEPA. Drinking Water Regulations; Maximum Contaminant Level Goals and National Primary Drinking Water Regulations for Lead and Copper; Proposed

low: 0 - 0.5; moderate: 0.5 - 1.0; and high: >1.0. Table 6.10 lists important risk assessment components, and the relative magnitude and direction of their impact.

6.5.1 Toxicity

Toxicity reference values (Rfds) for arsenic, antimony and cyanide add a high degree of uncertainty. At the present time a reference dose has not been verified for arsenic, and it is not known whether the value will be above or below the previously verified value. There is low confidence in the toxicity data base used to establish the antimony Rfd, and an uncertainty factor of 1000 is used (USEPA, 1989c). As a result it is possible that use of the value will overestimate risk. For cyanide, the lowest Rfd of all cyanide containing compounds has been used. Based on the analysis in Chapter 5, iron cyanide complexes are the most likely to be formed. These are the most stable forms, and therefore least bioavailable and toxic. As a result, use of the lowest Rfd for cyanide complexes will likely overestimate the risk.

Dermal contact risks are an order of magnitude higher than for soil ingestion (Tables 6-4, 6-6). One of the key assumptions in this exposure route is to adjust oral Rfd and potency values for all chemicals by a factor of 0.05 to convert administered to absorbed doses. This assumption is purposefully conservative since there is little chemical specific information regarding absorption. In addition, there may be significant differences between oral absorption and dermal absorption in the degree to which a chemical is metabolized before it reaches the target organ. This adds greatly to the uncertainty associated with using oral Rfd and slope factors to estimate dermal risks.

6.5.2 Exposure

Another key assumption for the dermal route is that 0.1% or 1.0% of the material on the skin is absorbed. These assumptions are intended to be conservative, but little quantitative data has been developed regarding dermal absorption of inorganic contaminants to verify or improve these assumptions.

Since only one round of metals data could be used to estimate ground and surface water risks, maximum values were used. Antimony and lead were both shown by this analysis to pose potential risk. In both cases, the risk at the next highest value is significantly less. Due to the small sample size and difficulty in obtaining representative samples from monitoring well 1, use of maximum water concentrations from Round 3 may tend to overestimate the risk. Based on the difference between the

maximum and next highest value for lead and antimony, this may be as much as an order of magnitude.

6.6 SUMMARY

The human health risk from cyanide, arsenic and other contaminants is based on the likely future use of the site. Industrial (mining) activity is expected to be the most reasonable future use of the site, though there is currently little activity. Given this assumption, the most important exposure routes are ingestion of and dermal contact with soil, and ingestion of ground water or surface water. Exposure via inhalation of suspended particulates or volatile chemicals (HCN) is not expected to be an important exposure route.

Using reasonable maximum exposure assumptions, arsenic, antimony, and cyanide are the most important contaminants in water with respect to human health. Nitrate/nitrite and lead were each present in a single groundwater sample at concentrations above established criteria, though these values may not be representative of overall site conditions. Exposure to arsenic in water could result in an increased cancer risk of 2×10^{-4} . There is also a risk of noncarcinogenic effects, mainly neurologic, liver, and skin related, from arsenic, cyanide and other chemicals. The hazard quotient for these effects is 2.5.

The most important contaminant in soil is arsenic. Exposure to soil could result in an increased cancer risk of 2×10^{-3} . The hazard index of 2.4 indicates that soil exposure could also result in a risk of noncarcinogenic effects, principally skin and neurologic disorders.

Major components of the assessment which decreased the certainty of the results were the toxicity reference values used, assumed future land use, dermal contact pathway risks, and water data. Due to the uncertainty in these and other areas, conservative assumptions were made in order to be protective of human health. Therefore, cancer and noncancer risk estimates must be carefully interpreted. This is particularly important when evaluating noncarcinogenic effects where uncertainty factors of 2 - 3 orders of magnitude are used in dose-response assessment. Given this uncertainty and other conservative assumptions in the exposure assessment (eg. dermal pathway exposure assessment), exceeding a hazard index or quotient of 1.0 by up to a factor of 5 may not be significant, though this is clearly a subjective determination.

REFERENCES

- Agency for Toxic Substances and Disease Registry (ATSDR). 1989. Toxicological Profile for Arsenic. ATSDR/TP-88/02. March 1989.
- Bjorksten, J.A. Comp. Therapy 8: 73-76. 1982.
- Canadian Council of Resource and Environment Ministers. 1987. Canadian Water Quality Guidelines, Task Force on Water Quality Guidelines, Canadian Council of Resource and Environment Ministers. March, 1987.
- Craun, G.F., Greathouse, D.G., and Gunderson, D. H. 1981. Methemoglobin levels in young children consuming high nitrate well water in the United States. International Journal of Epidemiology, 10. pp. 309-317. As cited in IRIS.
- U.S. Environmental Protection Agency (USEPA). 1979. Water related environmental fate of 129 priority pollutants, volume 1. EPA-440/4-79-029a, 514p. 1979.
- Ibid. 1985. Drinking Water Criteria Document for Copper. Final Draft. Office of Drinking Water. EPA/600/x-84-190-1. March 1985.
- Ibid. 1986. Guidelines for carcinogen risk assessment. 51 Federal Register 33992, September 24, 1986.
- Ibid. 1987. Special Report on Ingested Inorganic Arsenic: Skin Cancer; Nutritional Essentiality. Office of Health and Environmental Assessment. SAB Review Draft. EPA/625/3-87/013A. November 1987.
- Ibid. 1988a. Drinking Water Regulations; Maximum Contaminant Level Goals and National Primary Drinking Water Regulations for Lead and Copper; Proposed Rule. 53 Federal Register 31516. August 18, 1988.
- Ibid. 1988b. Memo regarding arsenic risk evaluation.
- Ibid. 1988c. National Oil and Hazardous Substances Pollution Contingency Plan. Proposed Rule. 53 Federal Register 51395, December 21, 1988.
- Ibid. 1988d. National Primary Drinking Water Regulations. 40 CFR Part 141. July 1, 1988.

- Ibid. 1988e. Screening Procedures for Estimating the Air Quality Impact of Stationary Sources (Draft). Model: SCREEN, version 1.1, 88300. EPA 450/4-88-010. August 1988.
- Ibid. 1989a. Exposure Factors Handbook. Final Report. EPA/600/8-89/043. March 1989.
- Ibid. 1989b. Health Effects Assessment Summary Tables, Third Quarter FY 89. OERR 9200 6-303-(89-3). July 1989.
- Ibid. 1989c. Integrated Risk Information System (IRIS), online database. Antimony. July 1989.
- Ibid. 1989d. Arsenic.
- Ibid. 1989e. Cobalt.
- Ibid. 1989f. Lead.
- Ibid. 1989g. Molybdenum.
- Ibid. 1989h. Personal communication. Herman Gibb, Office of Health and Environmental Assessment. August 29, 1989.
- Ibid. 1989i. Personal communication. Jeff Cohen, Office of Drinking Water. October 24, 1989.
- Ibid. 1989j. Memorandum from: Henry L. Longest II, Director, Office of Emergency and Remedial Response; To: Directors, Waste Management Division, Regions I, II, IV, V, VII and VIII et. al.; Subject: Interim Guidance on Establishing Soil Lead Cleanup Levels at Superfund Sites. September 7, 1989.
- Ibid. 1989k. Memorandum from: John C. Chamberlin, Director, Office of Administration and Michael B. Cook, Director, Office of Drinking Water; To: Regional Administrators, et al.; Subject: Monitoring Drinking Water for Lead at EPA facilities.
- Ibid. 1989l. Memorandum from: Jonathan Z. Cannon, Acting Regional Administrator; To: Regional Administrators and Assistant Administrators; Subject: October 26, 1989 Revisions to sections in Final National Contingency Plan.
- Ibid. 1989m. Region 10 Cancer Risk Model. Environmental Services Division, Health and Environmental Assessment Section. October, 1989.

Ibid. 1989n. Region 10 Exposure Parameters. Environmental Services Division, Health and Environmental Assessment Section. October, 1989.

Ibid. 1989o. Region 10 Rfd Model. Environmental Services Division, Health and Environmental Assessment Section. October, 1989.

Ibid. 1989p. Risk Assessment Guidance for Superfund. Human Health Evaluation Manual, Part A. Interim Final. OSWER Directive 9285-701A. July 1979.

Woodward-Clyde Consultants, 1987. Community relations plan for the Silver Mountain Mine Superfund site, Okanogan County, Washington: Woodward-Clyde Consultants, San Francisco, California, 16 p.

CHAPTER 7 - BASELINE ENVIRONMENTAL EVALUATION

This chapter discusses the assessment of toxicity of contaminants, exposure of these contaminants to vegetation and wildlife, and the likely risk to biota that utilize the site.

7.1 CONTAMINANT IDENTIFICATION

Water and soil sample results were compared with reference values to identify contaminants to be evaluated in the ecological risk assessment. The sample results used were the same ones selected for the human health assessment. The screening revealed seven metals that were examined further. This consisted of an in-depth evaluation of the screening criteria and data upon which the criteria were based.

Following the initial screening using reference values, water samples were evaluated for any toxicity as drinking water. No toxic constituents were identified.

7.2 TOXICITY ASSESSMENT

7.2.1 Initial Screening

Surface waters were evaluated for toxicity to aquatic biota using it as habitat and to wildlife using it for drinking water. Ground water was evaluated for potential toxicity to aquatic biota exposed to the ground water upon its arrival at the nearest surface water body (Lake Aeneas). Reference values used to evaluate toxicity to aquatic biota were the Ambient Water Quality Criteria and data from AQUIRE (Aquatic Toxicity Information and Retrieval Data Base, managed by the USEPA Environmental Research Laboratory, Duluth, MN). Reference values used to evaluate toxicity to wildlife via drinking water, were derived from various sources (cited in the text).

Soil concentrations were compared with values derived from Environmental Profiles and Hazard Indices for Constituents of Municipal Sludge (EPA series produced by the Office of Water, Regulations and Standards; US Environmental Protection Agency, 1985a) and from reference sources and on-line searches of OHM/TADS (Oil and Hazardous Materials/Technical Assistance Data System) and RTECS (Registry of Toxic Effects of Chemical Substances) databases as accessed through CIS (Chemical Information Systems, Inc.) as well as IRIS (Integrated Risk Information System).

7.2.1.1 Evaluation of surface and ground water data

Of thirty-four dissolved constituents measured in water samples, nitrite was not evaluated due to lack of suitable reference data. Cyanide was reported as both total and weak acid soluble; only total cyanide was evaluated. For the remaining 32 constituents, concentration averages and maxima were evaluated for the stock tank, the seep, and ground water. When only one measurement was available for a particular constituent, it was evaluated as an average. A dilution of 800 was applied to ground water sample results to estimate concentrations where ground water would contact surface waters. This estimate does not consider the influence of background ground water concentrations. Undiluted ground water values were also compared with reference values, but only diluted values were evaluated further because the ground water table is well below the root zone of plants in the sagebrush community.

Water as habitat

For 19 constituents, water sample results were compared with water quality criteria (Appendix N, Table 1; US Environmental Protection Agency, 1985-1987). Toxicity information on the remaining 13 constituents was obtained where possible from other reference sources and on-line searches of AQUIRE (Appendix N, Table 2). Data for these were summarized (Appendix N, Tables 2 & 3) by group of organisms (amphibian, fish, invertebrate, and plant) as well as endpoint (lethal and toxic effects). Toxicity test exposure concentrations were used directly for initial comparisons. The lowest toxic and lethal concentrations were used. No aquatic toxicity information was obtained for tin.

Based on these initial comparisons, ground water reaching surface waters, following the 800-fold dilution, exhibited no toxicity. The constituents in the ground water do add some incremental toxicity to potential background toxicity, however, and this is discussed in the uncertainty section below.

Surface water, on the other hand, exhibited some toxicity based on the screening criteria. Concentrations of aluminum, arsenic, copper, iron, and lead exceeded water quality criteria (Appendix N, Table 1; note that results for these metals are based on only one measurement). Only aluminum and copper in the seep exceeded the criteria for acute effects. Chronic toxicity criteria were exceeded in (1) seep water by aluminum, copper, iron, and lead, and in (2) stock tank water by arsenic. Chronic toxicity of silver could not be evaluated since the detection limit of 5.0 $\mu\text{g/L}$ was well above the chronic criterion of 0.12 $\mu\text{g/L}$. Magnesium and sodium occurred in toxic amounts in both the seep and the trough (again, based on only one

measured value for each metal; see Appendix N, Table 4).

Water as drinking water

Before constituents were evaluated for toxicity as drinking water, they were screened for toxicity using water quality criteria and the AQUIRE toxicity reference values. It was assumed (and subsequently substantiated, see next paragraph) that evaluation of the toxicity of water as habitat (which includes multiple routes of entry to an organism) is much more conservative than evaluating the toxicity of water as drinking water (only an oral route of entry). Based on the initial screening, aluminum, arsenic, copper, iron, lead, magnesium, and sodium appeared to occur in toxic concentrations and so were examined below in further detail.

These constituents were evaluated using data on toxic doses obtained from National Academy of Sciences documents, Eisler (1988), and other documents and databases identified in 7.2.1. Toxic concentrations were estimated from doses using the assumptions and method presented in Appendix O. No constituents were considered to be present in concentrations toxic to organisms that might drink the water in the stock tank or the seep. In general, these waters were from 1 to 4 orders of magnitude less toxic as drinking water than as habitat.

Aluminum

A single, relatively high concentration of aluminum (2999 $\mu\text{g/L}$) was measured in the seep water. Lowest oral LD50s for the rat and mouse are 3730 mg/kg and 1623 mg/kg (Appendix N, Table 5). These correspond to toxic concentrations of 37,300 and 16,230 $\mu\text{g/L}$, approximately 4 orders of magnitude above the seep concentration.

Arsenic

Arsenic toxicity was evaluated two ways, first assuming all the arsenic was in the trivalent form, and then assuming all was in the pentavalent form. The pentavalent form (arsenate) is expected to predominate (see 5.3.3.1.). The water quality criterion for acute arsenic (pentavalent form) toxicity was not exceeded. The criterion for chronic toxicity, however, was exceeded by a factor of 18, but only in the stock tank.

Data on wildlife toxicity to arsenic were obtained from National Academy of Sciences (1977a) and Eisler (1988). Sensitive bird species include the turkey with an LD50 dose of 17.4 mg/kg body weight (phenylarsonic acid). Other sensitive

bird species have died following single oral doses of 47.6 mg As/kg body wt. The mallard has an LD50 of 323 mg/kg (sodium arsenite). Mammals have been shown to be susceptible to as little as 5 mg As/kg in diet or single doses of 2.5-33 mg As/kg body weight. White-tailed deer were killed in an area of apparent misuse of arsenic acid used to control Johnson grass where soils contained approximately 2.4 mg As/kg, and water contained 0.42 mg As/L. Rabbits have died after receiving single doses from 8 to 40.4 mg/kg body weight depending on the form of arsenic.

The lowest lethal dose of 2.5 mg/kg corresponds to a water concentration of 25000 $\mu\text{g/L}$. Even the above-cited value of 420 $\mu\text{g/L}$ implicated in deaths of white-tailed deer is well above the concentration of concern (95 $\mu\text{g/L}$) measured in the stock tank.

Copper

The concentration of copper in a single sample taken from the seep only slightly exceeded the screening criterion for acute toxicity and was only 1.6 times the criterion for chronic toxicity. Toxic concentrations for ruminants range from 20-50 mg/L and exceed 250 mg/L for swine and rats (National Academy of Sciences, 1977b). These correspond to concentrations of 20,000-250,000 $\mu\text{g/L}$ which are 3 to 4 orders of magnitude above the 19 $\mu\text{g/L}$ measured in the seep.

Iron

The concentration of iron in a single sample taken from the seep exceeded the screening criterion for chronic toxicity by a factor of 5. Iron concentrations above 500 $\mu\text{g/g}$ dry weight in feed have been associated with poorer weight gain and food consumption in cattle (EPA Environmental Profile and Hazard Index for Iron). Assuming 40% of body weight is ingested daily (EPA Environmental Profile and Hazard Index for Lead, Table 4-4, footnote f: daily food intake/body weight ratio of 2 kg/5 kg for young cattle), this dose is approximately 200,000 $\mu\text{g/kg}$:

$$(40\%[\text{kg feed/kgBW}] \times \text{BW} [\text{kgBW}] \times 1000 [\text{g/kg}] \times 500 [\mu\text{g/g feed}]) / \text{BW} [\text{kg}]$$

This corresponds to a concentration of 2,000,000 $\mu\text{g/L}$, which is 400 times greater than the seep concentration of 4881 $\mu\text{g/L}$.

Lead

The concentration of lead in a single sample taken from the seep (7.1 $\mu\text{g/L}$) exceeded the screening criterion for chronic

toxicity by a factor of 2.2. Data on lead toxicity to domestic animals and wildlife were obtained from the EPA Environmental Profile and Hazard Index for Lead. The LC50 for chickens is 320 mg/kg dry weight (DW). Ducks suffer mortality at intakes greater than 8 mg/kg (DW). Swine exhibit decreased growth and feed intake as well as muscle tremors at 33 mg/kg (DW) and mortality at 66 mg/kg (DW). A different study, however, found only hypersensitivity at 86 mg/kg (DW). Horses appear to exhibit toxic effects at 1.7 mg/kg (DW) and mortality at an estimated dose of about 7 mg/kg (DW). The lowest value of 1.7 mg/kg corresponds to a concentration of 17,000 $\mu\text{g/L}$ which is over 3 orders of magnitude greater than the seep concentration.

Magnesium

The concentration of magnesium in single samples taken from the seep and stock tank only slightly exceeded the screening value. Only two toxicity reference values were available. The value used is the LC50 for a freshwater amphipod crustacean. The other effect is for a concentration 3 times greater that caused mortality in bluegill. Toxicity of magnesium is of low concern. It is an essential nutrient for which ambient water quality criteria have not been developed for drinking water or protection of aquatic life.

Sodium

The concentration of sodium in single samples taken from the seep and stock tank was about 8 times the screening value. The sodium toxicity value used is for an effect on enzymatic activity of the white sucker. The next lowest concentration to affect fish (the enzymatic activity of the mosquito fish) is an order of magnitude greater. Sodium is biologically essential and normally considered non toxic. The seep and tank values are around 16,000 $\mu\text{g/L}$. These are below the 20 mg/L suggested guidance level for human exposure to sodium in drinking water for individuals at risk of developing hypertension (USEPA 1985b).

7.2.1.2 Evaluation of soil data

Twenty-seven constituents were measured in soil samples. Cyanide was reported as both total and weak acid soluble, only total was evaluated. For 14 of the constituents, the soil concentrations were compared with values derived from Environmental Profiles and Hazard Indices for Constituents of Municipal Sludge.

Reference soil concentrations ($\mu\text{g/g}$ dry weight) were

obtained for toxicity to: (1) soil biota, (2) plants (values for 1 and 2 taken directly from the sludge documents), (3) predators consuming soil biota, and (4) herbivores consuming plants grown on contaminated soil (3 and 4 are derived values) (Appendix N, Table 6). These four concentrations were compared with maximum and average concentrations for the soil samples (Appendix N, Table 7).

Of the remaining 12 constituents, calcium, magnesium, potassium, and sodium were not analyzed further, due to their essentiality and general lack of toxicity. Toxicity information for the remaining 8 pollutants was obtained where possible from reference sources and OHM/TADS (Oil and Hazardous Materials/Technical Assistance Data System) and RTECS (Registry of Toxic Effects of Chemical Substances) databases as accessed through CIS (Chemical Information Systems, Inc.) as well as IRIS (Integrated Risk Information System). These data are summarized in Appendix N, Table 8.

The lowest (i.e., most conservative) toxic dosages (LD50s for example) were converted to soil concentrations by estimating the amount of soil ingested by each particular organism. These estimates were derived (as explained in detail in Appendix O) based on assumptions used in the ecological risk assessment for the Crab Orchard superfund site (Clark, 1989) and are summarized in Appendix N, Table 9. These concentrations were then compared with on-site values (Appendix N, Table 10).

Based on the initial screening (Appendix N, Table 10) nine constituents were evaluated further to determine the location, frequency, and magnitude by which the soil values exceeded the reference values. These constituents were those for which the ratio of the average concentration to the reference value exceeded 1 or the ratio of the maximum concentration to the reference value exceeded 2. Appendix N, Table 10, summarizes these findings, and shows that arsenic, cadmium, manganese, and zinc are of very high concern throughout the site, followed by selenium which is of medium concern. All but selenium exhibited a pattern of increasing toxicity in the heap and dump samples over background samples. However, manganese, selenium, and zinc all had toxic background values as well. Copper and lead values exhibited high toxicity in the south toe of the heap.

7.2.2 Toxicity of surface and ground water

Based on initial screening, surface and ground water at the site are not expected to be toxic to plants or animals.

7.2.3 Future toxicity of cyanide and arsenic in ground water

Predictions in Chapter 5 suggest that future leachate may contain up to 2.2 mg/L cyanide (likely) and 27 mg/L arsenic (probable, with the passage of tens of years). Following a dilution of 800, these concentrations, as the ground water enters Aeneas Lake, could be 2.8 $\mu\text{g/L}$ cyanide and 34 $\mu\text{g/L}$ arsenic. The chronic water quality criteria for cyanide and arsenic are 5.2 $\mu\text{g/L}$ and 48 $\mu\text{g/L}$, respectively (assuming conservatively that all the arsenic is in the more toxic pentavalent form). These criteria are not likely to be exceeded in Aeneas Lake because the initial mixing of lake and ground water will dilute the concentrations.

7.2.4 Toxicity of soil

Based on the initial screening, arsenic, cadmium, manganese, selenium, and zinc concentrations were of concern throughout the site. Copper and lead values exhibited toxicity in the south toe of the heap.

Arsenic, cadmium, manganese, and zinc (as well as lead and copper on a more limited scale) appear to have increased in certain soils due to the mining activity.

For the mine dump and nearby soil, existing toxicity was very similar in frequency and magnitude to that of the potential toxicity from the heap if the cover were removed. In particular, should the cover fail, it is the south toe of the heap that appears to have very high or the highest concentrations of many metals of concern.

Arsenic

Arsenic concentrations in the soil are of concern. As discussed below, they are high enough to be toxic to vegetation and ruminants. The value used for screening was 45 $\mu\text{g/g}$ (or mg/kg); a lower value, protective of vegetation, has been recommended by Eisler (1988).

Depressed crop yields have been recorded for arsenic levels at 3 to 38 mg of water soluble soil As/L. Arsenic availability to plants is highest in coarse-textured soils having little colloidal material and little ion exchange capacity (and low organic matter). To be absorbed, the arsenic compounds must be in a mobile form. In general, soil microorganisms are capable of tolerating and metabolizing relatively high concentrations of arsenic. To achieve no observable effects in vegetation, the

criteria of <1 mg total water soluble soil As/L, <25 mg total As/kg soil, and <3.9 μg As/m³ air are recommended (Eisler, 1988).

Mammals may be affected by arsenic levels. White-tailed deer have been killed in an area of apparent misuse of arsenic acid used to control Johnson grass (soils in this area contained approximately 2.4 mg As/kg; Eisler, 1988). Eisler (1988) suggests that cattle may develop a preference for weeds sprayed with arsenic because of the saltiness and that small mammals may avoid arsenic-treated foods if given a choice.

The probability of chronic arsenic poisoning from continuous ingestion of small doses is rare, because detoxication and excretion are rapid (Eisler, 1988). For most species of mammals, arsenic trioxide from 3 to 250 mg/kg body weight or sodium arsenite from 1 to 25 mg/kg body weight is lethal.

In some areas, high arsenic concentrations appear to be innocuous to animals due to antagonism by selenium (National Academy of Sciences, 1977a). Soil analyses show that selenium was widespread throughout the site. Whether any antagonistic interaction with arsenic occurs here is unknown.

Cadmium

Based on the screening concentrations, cadmium exhibited possible toxicity to plants (decrease in lettuce, wheat, and soy yields) and a more likely toxicity to predators (decreased egg production in chickens) feeding on soil biota (earthworms; which have a fairly high rate of uptake of cadmium from soil). It is extremely difficult to extrapolate these uptakes and effects to a sagebrush ecosystem, so the database is examined here further. While some toxicity to wildlife and vegetation is possible, it is likely to be minimal.

Some data exist describing the toxicity of cadmium to plants. Most of the endpoints are decreased yield for common vegetables. The range of values for 9 tests in which no effects were observed was 5 to 640 $\mu\text{g/g}$, with a geometric mean of 27 $\mu\text{g/g}$. Toxic effects, however, appeared at concentrations as low as 2.5 $\mu\text{g/g}$. The range of values for 14 test in which effects were reported at a level of at least 50% was 10 to 200 $\mu\text{g/g}$, with a geometric mean of 93 $\mu\text{g/g}$. Average cadmium values only exceeded 2.5 $\mu\text{g/g}$ in the south toe (where the average was 3.0 $\mu\text{g/g}$), so toxicity to plants is likely to be very limited. This conclusion assumes that uptake of cadmium by plants on site and resultant effects, are similar to those for vegetables. Only limited data address uptake of heavy metals in sagebrush. These data suggest that uptake is slow, so utilizing the vegetable data

may represent a conservative evaluation of plant toxicity.

The data on uptake rates of cadmium by soil biota are for earthworms. The rates from 4 studies ranged from 0.64 to 13.7 with a geometric mean of 2.4. The worst-case of 13.7 was used in the screening. The data on toxicity of cadmium to domestic animals and wildlife indicate that mallards suffer kidney tubule degeneration on a diet with cadmium in the feed at a concentration of 200 $\mu\text{g/g}$. Japanese quail undergo various toxic effects such as decreased body weight when fed concentrations of 75 $\mu\text{g/g}$. Slight toxicity in rats is observed when they are fed concentrations of 31 to 45 $\mu\text{g/g}$. Chickens exposed over the course of 12 weeks, exhibit toxic effects for concentrations from 3 to 48 $\mu\text{g/g}$; however this same study showed that over the course of 48 weeks, that no effect occurred at 3 $\mu\text{g/g}$.

Quail and mallards may be more representative of on-site wildlife. If the quail toxicity data were used, the reference value would be 5.47 which is greater than any of the average cadmium concentrations on site.

Copper

Copper concentrations are at toxic levels only in the south toe of the heap. The reference values that were exceeded were 100 $\mu\text{g/g}$ for toxicity to plants and 131 $\mu\text{g/g}$ for toxicity to soil biota. The data used (Sludge document for copper) to select these values were examined further. As discussed below, no compelling data were available to suggest the reference values be revised.

Limited data were available on effects to soil biota (six studies of earthworms and one of soil bacteria cultured in a liquid medium). The next lowest concentration exhibiting toxic effects on earthworms was an LC50 of 150 $\mu\text{g/g}$.

The database summarized approximately 60 studies, mostly concerning effects on vegetables and grains. Toxic effects of copper on corn have been shown to occur at concentrations as low as 45 $\mu\text{g/g}$ DW; but at this level, copper can also increase yields of rye and corn. The screening level of 100 $\mu\text{g/g}$ is a level above which rye, corn, and wheat are consistently affected by copper.

Lead

Lead concentrations are at toxic levels mainly in the south toe of the heap, with some toxicity in the mine dump. The reference values that were exceeded were 100 $\mu\text{g/g}$ for toxicity to

plants and 86 $\mu\text{g/g}$ for toxicity to predators on soil biota. The data used (Sludge document for lead; US Environmental Protection Agency 1985a) to select these values were examined further. As discussed below, the plant toxicity value is highly conservative, but the choice of a more reasonable value is not obvious. The toxicity to soil biota predators appears to be of low concern if mallard and quail data are considered more realistic than the most conservative value.

Over thirty studies of phytotoxicity were summarized in the database. Soil concentrations of lead as high as 1000 $\mu\text{g/g}$ have been shown to have no effect on lettuce or oat yield, depending on the existing tissue concentration prior to application of the lead. Wheat, oats, and radish roots show decreased biomass at 1000 $\mu\text{g/g}$. Peanuts were not adversely affected at 820 $\mu\text{g/g}$, while alfalfa in some studies showed no reduction in yield at 100 $\mu\text{g/g}$. One study of corn found increased yield at 186 $\mu\text{g/g}$. On the other hand, some studies showed that yield of oats and red clover is reduced when soil concentrations exceed 50 $\mu\text{g/g}$. The soil concentration of 100 $\mu\text{g/g}$ was selected in the sludge document as a "worst" case. Given the wide range of concentrations that may or may not exhibit toxicity, the reference value will be retained at this time.

Toxicity to predators on soil biota is a function of uptake by soil biota and feed concentration toxic to a predator. The highest uptake rate of 0.54 (tissue/soil) was used. Two other slope values were presented in the database were not much different (0.41, for the woodlouse; and 0.33, a different study of earthworms), so the reference slope is retained. The feed concentration of 46 $\mu\text{g/g}$ corresponds to the potency of contaminated feed to bring about a daily intake dosage of 8 mg/kg lethal to ducks (not identified to species). A review of the database revealed that quail can withstand 2000 $\mu\text{g/g}$ lead in feed without adverse effect but suffer depressed body weight and increased mortality at 3000 $\mu\text{g/g}$. Mallards suffer mortality at 200 $\mu\text{g/g}$. The duck data are the most conservative. Using the mallard value of 200 $\mu\text{g/g}$ instead of the duck value of 46 $\mu\text{g/g}$, the reference concentration for the soil becomes 370 $\mu\text{g/g}$, which is above the maximum site value.

Manganese

The preliminary screening (using a soil concentration of 194 $\mu\text{g/g}$) indicated that manganese is toxic throughout the site (including background stations), particularly at the south toe of the heap where the average concentration is about twice that at other locations. This screening value was based on the lowest lethal dose, which was for dogs. Screening based on doses for

other animals suggests manganese will not be toxic to rodents, but could be to lagomorphs. The National Academy of Sciences, 1973 document on manganese was reviewed to evaluate toxicity to plants and animals. Based on this review, as discussed below, phytotoxicity may be present but is likely to be ameliorated by the existing iron content. Toxicity to ruminants is difficult to predict, but may not be present.

According to the National Academy of Sciences (1973), manganese availability in soil is closely related to pH and oxidation-reduction potentials. Manganese absorption by plants is a function of temperature and light intensity. Toxicity to plants can be reduced by increasing the concentration of other cations such as calcium, zinc, copper, and magnesium that compete for absorption. Iron salts can reduce manganese toxicity in tobacco, rice, and corn. In soil, iron to manganese ratios from 1.25 to 3.08 are associated with no toxicity or only mild toxicity to plants. At the south toe of the heap the ratio of the mean concentrations of iron and manganese was 43, suggesting that phytotoxicity due to manganese may be of low concern. Data concerning toxicity to cereals (barley, oats, rice, rye, and wheat) indicates toxicity begins in the range from 80 to 2500 ppm ($\mu\text{g/g}$); lack of toxicity to cereals has been observed in the range from 14 to 783 ppm ($\mu\text{g/g}$). At low levels, plants suffer from manganese deficiency. Based on field studies, forage crops (e.g., alfalfa) have been shown to be subject to toxic effects at soil concentrations of 477 to 650 $\mu\text{g/g}$. The above data suggest some possibility of phytotoxicity due to manganese concentrations in soils on site, but there some evidence to suggest that the iron concentrations on site could reduce this potential.

Of the trace elements, manganese is among the least toxic to mammals and birds. Hens tolerate 1000 ppm without ill effects, but 4800 ppm is toxic to young chicks. Rats tolerate 2000 ppm; calves exhibit lower weight gain at 2460 ppm with no effect at 820 ppm. When manganese is high, lambs, cattle, rabbits, and pigs have problems maintaining hemoglobin concentrations. Using the value of 820 ppm for calves as a surrogate for deer and using the assumptions in Appendix O for deer ingestion rates of soil and vegetation grown on contaminated soil, the toxic soil concentration is calculated as 7,523 $\mu\text{g/g}$, approximately an order of magnitude above measured levels.

Selenium

Selenium appears to be the only pollutant that occurred in toxic concentrations throughout the site, but with no increase in the heap or dump. This suggests that mining activity did not elevate selenium concentrations, and that selenium may be

naturally high in this area. The selenium toxicity database was examined further to evaluate toxicity of on-site concentrations.

The preliminary screening (using a soil concentration of $0.45 \mu\text{g/g}$) indicated that selenium is toxic to herbivores throughout the site (including background stations), particularly at the south toe of the heap and at the dump. Some phytotoxicity was suggested in the dump and at background stations, based on a soil concentration of $1.0 \mu\text{g/g}$. An evaluation of selenium toxicity (see below) suggests that selenium may not be phytotoxic. Rodents and birds (but not ruminants), however, may be at risk from soil toxicity.

Results from 9 studies of phytotoxicity indicate toxic selenium concentrations in soil range from 1.12 to $5 \mu\text{g/g}$ with a mean of $2.68 \mu\text{g/g}$. The maximum concentration measured on site was $2.0 \mu\text{g/g}$, suggesting that selenium phytotoxicity may be of low concern.

Toxicity to herbivores is a function of uptake by plants and plant concentrations toxic to herbivores. A detailed evaluation of uptake data is presented in the sludge document for selenium, pages 3-6 to 3-7. Maximum uptake slopes ($\mu\text{g/g}$ tissue DW divided by kg/ha) are listed for wheat (7.8), alfalfa (2.6), and corn (5.2). Using the uptake rate of 7.8 and an on-site selenium concentration of concern of $1.23 \mu\text{g/g}$ (ave concentration at the dump), the feed concentration of concern is estimated to be $19 \mu\text{g/g}$. Based on 22 studies of toxicity in chickens, toxicity occurred when feed concentrations exceeded $5 \mu\text{g/g}$, but a weight gain was observed for concentrations between 5 and $40 \mu\text{g/g}$ for selenium fed as SeO_2 (4 studies). Based on 8 studies, rats were affected (chronic poisoning) by feed concentrations as low as $1 \mu\text{g/g}$. However, one study in which rats were fed seleniferous wheat found only a slight decrease in weight gain at concentration of $4.4 \mu\text{g/g}$ (moderate toxicity occurred at $8.8 \mu\text{g/g}$, and marked toxicity at $17.5 \mu\text{g/g}$). A hamster study found effects on weight gain at $9 \mu\text{g/g}$. Nine studies indicate a mixed response by swine to selenium concentrations. Some studies suggest that swine may tolerate 5 to $10 \mu\text{g/g}$, while other studies indicate toxic effects occur at $7 \mu\text{g/g}$. Horses suffer chronic poisoning at $44 \mu\text{g/g}$ and mortality at $115 \mu\text{g/g}$. A study of cattle, sheep, and horses found subacute selenosis when forage contained concentrations from 20 to $50 \mu\text{g/g}$. These data suggest that rodents and birds may be at risk, but that ruminants are not.

Zinc

Zinc appeared to occur in toxic concentrations throughout

the site with highest concentrations in the heap and lowest concentrations in the background samples. This suggests that mining activity elevated zinc concentrations even though zinc may be naturally high in this area. The zinc toxicity database was examined further to evaluate toxicity of on-site concentrations.

The preliminary screening (using a soil concentration of 42 $\mu\text{g/g}$) indicated that zinc is toxic to predators on soil biota throughout the site (including background stations), particularly at the south toe of the heap. Phytotoxicity is likely in the heap based on a soil concentration of 224 $\mu\text{g/g}$, and toxicity to herbivores is likely based on a concentration of 211 $\mu\text{g/g}$. An evaluation of zinc toxicity (see below) suggests the heap may be toxic to birds, possibly toxic to plants, and perhaps not toxic to herbivores.

Toxicity to predators on soil biota is a function of uptake rates by the biota and food concentrations toxic to predators. Available studies on uptake indicate a range for earthworms (0.078 to 2.95 with a geometric mean of 0.51, $n=4$) and a value of 0.28 for the woodlouse. Use of the geometric mean would raise the reference value for toxicity to predators to 243 $\mu\text{g/g}$, which still indicates the presence of zinc toxicity in the heap. Data on toxic concentrations of food indicate toxic effects to quail at 125 $\mu\text{g/g}$ and lethal effects at 2000 $\mu\text{g/g}$. The maximum tolerable level for poultry is given as 1000 $\mu\text{g/g}$. Turkeys experience decreased growth at 4000 $\mu\text{g/g}$; chickens can exhibit decreased growth at 1500 $\mu\text{g/g}$. These data suggest toxicity to birds is likely, although doses may not be lethal.

Lab studies of the phytotoxicity of zinc indicate that corn yield is affected at concentrations above 240 $\mu\text{g/g}$ (10 studies), whereas a field study found no toxicity at 606 $\mu\text{g/g}$. Other studies utilizing pots, found toxicity to wheat beginning at 384 and toxicity to lettuce beginning at 224 $\mu\text{g/g}$. These data indicate that phytotoxicity is possible on site.

Toxicity to herbivores is a function of uptake by plants and toxic concentrations in food of herbivores. Approximately 60 uptake slopes have been calculated for a variety of vegetables and grains. These slopes range over 3 orders of magnitude, from 0.005 to 3.97 ($\mu\text{g/g}$ plant tissue divided by kg/ha). One study, of vegetation in a field subject to fallout from a nearby smelter, yielded a rate of 0.064. Rates for grains (oats, corn, barley, etc.) which might be more representative of on-site plants, range from 0.005 to 0.97 with a geometric mean of 0.084 ($n=17$). Using this rate (geometric mean), rather than the screening level of 0.71 and retaining the concentration of food toxic to herbivores of 300 $\mu\text{g/g}$, the screening level for soil

toxicity to herbivores would be about 1800 $\mu\text{g/g}$, well above any measured value. This suggests that toxicity may not be of concern on site.

The maximum tolerable level for cattle and horses is given as 500 $\mu\text{g/g}$; for sheep, 300 $\mu\text{g/g}$; and for swine, 1000 $\mu\text{g/g}$. Swine can exhibit toxic effects at 268 $\mu\text{g/g}$ (zinc lactate in milk) or no adverse effect at 500 to 1000 $\mu\text{g/g}$ (ZnCO_3). The toxic concentration of zinc in feed of 300 $\mu\text{g/g}$ appears reasonable to use in evaluating toxicity to herbivores, although ruminants may tolerate higher concentrations.

The interactive effect of copper with zinc on site is unknown. Copper is known to be prophylactic in rats against anemia and other effects in the liver associated with zinc toxicosis (National Academy of Sciences, 1977b).

7.2.5 Toxicity tests and their quality

Water pollutant reference values are the water quality criteria. Water quality criteria are promulgated only after a screening and derivation process that includes evaluation of wildlife toxicity data. Similarly, toxicity tests are entered into the AQUIRE database after meeting screening criteria.

Of all the pollutants measured in surface water, ground water (before dilution was applied), and soil, only lead and copper exhibited toxicity in all three types of samples. In general, toxicity for a given pollutant was not consistent across the sample types.

Soil metals reference values were obtained from the EPA sludge documents. These values are based on the most sensitive species. Most values are based on plants (mainly vegetables) grown on sludge, domesticated animals feeding on these plants, biota such as earthworms living in the soil, and domesticated birds such as chickens and turkeys feeding on the soil biota. The most conservative values available from conventional toxicity tests as reported in the searched databases were used. The quality of these data are addressed somewhat in several of the databases. Much of the information is peer-reviewed (except for RTECS data) or undergoes a screening process before entry into the data base.

7.2.6 Uncertainty

The ground water from the site is predicted to undergo a

dilution of 800 or more. This dilution is predicted where the ground water from the mine site mixes with the main aquifer prior to reaching Aeneas Lake. No further dilution is assumed upon reaching the lake. The dilution water, however, has certain levels of constituents found in the on-site ground water. The on-site ground water therefore, will add to the constituents in the dilution water. Concentrations in the off-site irrigation wells were used as surrogate measures of concentrations in dilution water. For most constituents, this increment is less than one percent of the reference value for diluted ground water (see Appendix N, Tables 1 & 4). Each constituent in undiluted ground water whose concentration exceeded the reference value was examined further. The incremental toxicity from the site was added to the background to determine whether the resulting water exceeded the reference value. As shown in Appendix P, the on-site increment did not cause any exceedances in the receiving water. In the cases of copper, lead, and sodium, the increment added to an existing background level that exceeded the screening levels.

All of the stock tank and seep concentrations exhibiting toxicity were single values. Without replication, uncertainty in evaluating toxicity is high since there is no estimate of variability, maxima, or averages.

The dilution estimate of 800 for ground water entering the closest surface water is somewhat uncertain. If dilution is assumed to be an order of magnitude less, the effect on the toxicity assessment can be determined by examining the ratios of diluted ground water to the reference values as shown in Appendix N, Tables 1 & 4. By moving the decimal one place to the right, only aluminum has a ratio that exceeds one (in this case, the ratio compares the maximum ground water value with the water quality criterion for chronic effects). Cyanide and sodium have the next highest ratios, which, under this lower dilution, would be 0.7. For cyanide, the dilution water concentration is assumed to be 0 (based on samples collected from off-site wells), so no toxicity would be expected in the surface water. For sodium, the dilution water concentration is estimated to be 15,548 $\mu\text{g/L}$. When divided by the reference value of 2,000 $\mu\text{g/L}$, the resultant ratio is 7.77 and the site ground water would be expected to add 0.7 to this ratio (i.e., a contribution of about 9%). If the next lowest reference value were used, the total toxicity from dilution water and the on site ground water would be about 0.85. Based on the above analyses, it is likely that, apart from aluminum toxicity, that a large decrease in the estimate of dilution would not result in increased estimates of toxicity.

A major source of uncertainty is in extrapolating dose-

response relationships determined under laboratory conditions and with laboratory species to naturally occurring plants and wildlife. An additional large category of uncertainty involves the assumptions made to estimate dosages of soil and water by wildlife.

Confidence in the toxicity tests increases as more tests are completed on a wider variety of organisms. Appendix N, Table 11, indicates the breadth (# of groups of organisms) and frequency (# of test results) of the available toxicity data. Appendix N, Tables 2 & 5, can be consulted for more details on the type and amount of toxicity data used in this toxicity assessment.

7.3 EXPOSURE ASSESSMENT

7.3.1 Potentially exposed populations

To document the important ecological characteristics of the site and to evaluate the effects on the local ecosystem, EPA staff visited the site June 7 and 8, 1989. The visit included observation of the plant community and investigation of nearby surface waters.

To complement site observations, representatives of the Forest Service (Tonasket-George Halekas, Bill Randall), Department of Wildlife (Ron Friesz, Jerry King, Dale Svedburg), and the Natural Heritage Program (Rex Crawford) were contacted for information about eastern Washington sagebrush ecosystems and site-specific natural history. Some information about the soil and water samples collected at the site was obtained from Bureau of Mines personnel (Dave Denton, John Benham).

Six aerial photographs taken between 1952 and 1985 (US Environmental Protection Agency, 1989) were examined by an EPA plant toxicologist to determine whether major visible damage to vegetation had occurred near or at the site (Kapustka, 1989).

A literature search was conducted for additional information on sagebrush ecosystems in eastern Washington. The results of the site observations, literature search, and discussions with agency personnel are discussed below.

7.3.1.1 Ecosystem characteristics

Like many areas of Washington east of the Cascade Mountains, the Silver Mountain Mine site is part of a sagebrush ecosystem. This type of ecosystem has been studied extensively, particularly in the area around Hanford reservation, and is characterized by a

predominance of sagebrush and bunchgrass. Although winters at the Silver Mountain Mine are somewhat cooler and there is more precipitation than at Hanford (Crawford, oral communication), the areas share a number of features and are likely to support many of the same plants and animals.

Sagebrush (Artemesia tridentata) is the dominant shrub species in this ecosystem. Sagebrush leaves are food for mammals such as black-tailed jackrabbits (Lepus californicus), voles, and mule deer (Odocoileus hemionus). The plants provide nesting sites for sage sparrows (Amphispiza belli) and other small birds (Schwartz, 1983; Rickard and Van Scoyoc, 1984) as well as habitat and food for grasshoppers (Sheldon & Rogers, 1978). The roots of the sagebrush plants are concentrated mostly in the upper 0.5 m (Hanford site; Rickard and Van Scoyoc, 1984).

Trace elements in air emissions have been detected in sagebrush leaves following incorporation of these pollutants into the root zone. Rickard and Garland (1983) found that the same order of trace element abundance occurs between leaf and soil samples. Rates of uptake to the leaves may be slow, however. Rickard and Van Scoyoc (1984) found that two years after a one-time application of molybdenum to the soil surface, this element appeared in sagebrush leaf samples. This was not the case for copper, manganese, or zinc, however.

Bunchgrass is another major plant in the sagebrush community and indicates where soil is deeper and moisture more plentiful. Most wildflowers in the sagebrush ecosystem are annuals, sprouting in fall, growing on mild winter days, and flowering in the spring (Schwartz, 1983). Primary production is low compared with other kinds of wild plant communities in the semi-arid parts of North America. In the Hanford area sagebrush community, primary production by the herb stratum ranges between 10 and 195 g/m²/y (Rickard and Rogers, 1983). Despite low productivity, plants are utilized by seed-eaters (desert rats, harvester ants, pocket mice, etc.), shrub eaters (sagebrush voles, jackrabbits, etc.), and other herbivores (squirrels, yellow-bellied marmots, etc.) (Schwartz, 1983).

The primary natural disturbance to vegetation in the sagebrush ecosystem is wildfire. Sagebrush is slow to recolonize burned areas (Rickard & Rogers, 1983). The development of orchards (Carson and Peek, 1987) and other agriculture, as well as cattle grazing, have also caused the loss of much sagebrush ecosystem.

According to Crawford (oral communication), the occurrence of sagebrush and needle and thread grass (Stipa comata) in

combination (the sagebrush/needle and thread grass complex) is one of the most sensitive sagebrush complexes. The occurrence of Stipa as the major understory grass is indicative of an undisturbed sagebrush community. Because of the physical disturbance by land uses at the site, Stipa comata is unlikely to occur at the Silver Mountain Mine site in any significant abundance.

The dominant invertebrates in the sagebrush community by biomass are below-ground insects, soil mites, and nematodes (Schwartz, 1983). Insect larvae, which often burrow in the soil, are especially important in the diet of young birds, deer-mice, and lizards (Rickard and Rogers, 1983). Above ground, scorpions, grasshoppers, and beetles are common invertebrates (Rickard and Rogers, 1983; Schwartz, 1983). The grasshoppers roost on sagebrush at night and many species also roost during the day to avoid the high temperatures of exposed soil surfaces (Sheldon and Rogers, 1978). The biomass of black beetles is significant in sagebrush ecosystems, reaching levels up to 20 kg/hectare, a biomass that sometimes exceeds that of birds and mammals (Rickard and Rogers, 1983).

Lizards, rattlesnakes, yellow-bellied racers (Coluber constrictor), king snakes, and gopher snakes (Pituophis catenifer) are important reptiles of the sagebrush ecosystem. The racers constitute an important component of the diet of nesting Swainson's Hawks (Rickard & Rogers, 1983; Schwartz, 1983).

A variety of mammals inhabit sagebrush ecosystems. Rodents survive heat and drought by burrowing and living underground (Schwartz, 1983). The Great Basin Pocket-mouse (Perognathus parvus), at densities that can reach 56 mice/hectare, is probably the most abundant small mammal, along with deer-mice (Peromyscus maniculatus), harvest-mice (Reithrodontomys megalotis), Townsend ground squirrels (Spermophilus townsendii), and grasshopper mice (Onychomys leucogaster) (Rickard & Rogers, 1983). Jackrabbits, yellow-bellied marmots, and voles are also visible components of the small mammal community (Schwartz, 1983). Although small mammal abundance is too limited to affect the vegetal composition of the plant community, small mammals are an important food source for snakes, predatory mammals, and birds (Rickard & Rogers, 1983). Other mammals that may use the Silver Mountain Mine area include coyotes (Canis latrans), foxes, badgers (Taxidea taxus), and bobcats (Lynx rufus) (Rickard & Rogers, 1983; Schwartz, 1983).

Elk have been shown to utilize some sagebrush areas. McCorquodale (1987) has shown that during fall and winter in the

Hanford area, Rocky Mountain elk (Cervus elaphus nelsoni) females tend to rest or ruminate in areas of sagebrush and Sandberg's bluegrass (Poa sandbergii), even though they prefer to feed in areas where sagebrush has been removed by wildfires. Sagebrush also provides winter habitat for mule deer. Carson and Peek (1987) showed that adult female mule deer (Odocoileus hemionus) use of sagebrush habitat increases from 21% in summer to 35% in winter. Carson and Peek (1987) also point out that springs or seeps attract deer in summer (corroborating findings by Eberhardt, et al., 1984) and that sagebrush is used for cover rather than food. As noted in Section 2.7 and 3.3, there is a small seep on site and a trough outside the fenced area which collects mine drainage water.

Relatively few passerine birds (eg. perching and song birds) are supported by the sagebrush habitat. Nesting bird densities range from 130-260 per km² (Rickard & Rogers, 1983). However, coulees appear to support a variety of raptors. For example, the Esquatzel Coulee in southeastern Washington supports at least ten species of raptors, including Red-tailed (Buteo jamaicensis) and Ferruginous hawks (Buteo regalis) as well as Barn (Tyto alba) and Great Horned owls (Bubo virginianus) (Knight and Smith, 1982). Peregrine falcons and prairie falcons also occur in eastern Washington's arid areas (Schwartz, 1983). According to Halekas (oral communication), sensitive bird species in sagebrush ecosystems include the long-billed curlew (Numenius americanus), Swainson's hawk (Buteo swainsonii), and possibly the Columbian sharp-tail grouse. According to Svedburg (1989), additional species of concern include burrowing owls (Speotyto cunicularia) and golden eagles (Aquila chrysaetos). Other birds which sometimes occur in sagebrush habitats include western meadowlarks (Sturnella neglecta), horned larks (Eremophila alpestris), sage sparrows (Amphispiza belli), and desert swifts (Rickard & Rogers, 1983; Schwartz, 1983).

Sagebrush and bunchgrass characterize the area at and around the site. A single cottonwood tree is located on the site, and pine forests occur within a few miles. According to Bureau of Mines personnel, typical wildlife in the area include "deer, coyote, field mice, snakes, and various birds, including raptors" (Benham and Denton, 1989).

During the June 1989 site visit by EPA there was evidence (trails, droppings) that marmots were using the area of the mine dump. The U.S. Bureau of Mines (1988) has reported "the presence of snakes nesting beneath the cover" on the heap.

As noted in Section 2.10, the area around the Silver Mountain Mine site is used for grazing cattle, and several farms

exist in the area to the south. At the site itself roads, paths, mine dumps, and buildings, as well as trampling of the site by cattle watering at the trough, have disrupted the natural habitat.

In 1981, a calf was reported to have died after straying onto the site, and several birds were also found dead on site. The cyanide-contaminated leachate in the collection pond was neutralized after this, and in 1985 the heap and drained pond were covered with a plastic liner. No reports of animal deaths on site have been received since that time.

During the June 1989 site visit several species of birds were noted. Swallows were seen nesting inside the mine entrance, killdeer were observed adjacent to the mine dump, and robins were seen on site. In the coulee adjacent to the site, mourning doves (nesting) and shrikes were observed. At Stevens Lake to the north coots, ducks, and blackbirds were seen. Blackbirds were noted in the farming area several miles to the south. Such observations represent only a portion of the use of the area by birds, since the bird population varies with the season, the time of day, and availability of food and suitable habitat.

The species of concern in the Silver Mountain Mine area have been identified by the Washington State Department of Wildlife (King, 1989) and are listed in Table 7.1.

Aerial photos (6 photos from 1952-1985; US Environmental Protection Agency, 1989) were examined by an EPA plant toxicologist for assessment of any obvious damage to vegetation near and on the site (Kapustka, 1989). The only changes of note occurred in a "peninsular"-shaped area to the south of the tailings pile and these changes were attributed to possible differences in annual rainfall (Kapustka, 1989). Damage to vegetation however, occurs throughout the area around the heap due to cattle using the on-site trough.

7.3.1.2 Sensitive ecosystems

Several surface waters are available to wildlife in the area of the site. Surface water is discussed in Sections 2.7 and 3.3 and Figure 2.4 shows the locations of the nearest surface water bodies. Briefly, these include the shallow pool associated with the seep, the cattle trough, the mine drainage (the source of water for the trough), and seasonal pondings and marshy areas 0.5 miles east of the site in the Horse Springs Coulee.

Table 7.1 List of species of concern at the Silver Mountain Mine site in the Horse Springs Coulee area, as identified by the State of Washington, Department of Wildlife (King, 1989).

BIRDS

Golden eagle
Sharp-tailed grouse
Long-billed curlew
Sage sparrow
Lark sparrow
Song sparrow
Vesper sparrow
Mountain bluebird
Western bluebird
Horned lark
Tree swallow
Burrowing owl
Saw-whet owl
Harrier
Red-tailed hawk
Kestrel
Rough-legged hawk

REPTILES

Painted turtle

WATERFOWL

Mallard
Pintail
Shoveller
Widgeon
Redhead
Ruddy
Cinnamon teal
Blue-winged teal
Pied-billed grebe
Coot

MAMMALS

Mule deer
Coyote
Bobcat
Cottontail rabbit
Muskrat

The closest sensitive ecosystems are the small lakes and ponds to the north which are hydrologically upgradient from the site. Stevens Lake and the surrounding potholes and marsh area host nesting pairs of representative native species of waterfowl (King, 1989). The State of Washington, Department of Wildlife states that about 60 ducks can be observed in these areas from April to October (King, 1989).

The stock tank is of limited size and permanence. It is not considered an ecosystem even though it supports a limited, undescribed, aquatic biota. The seep is similarly limited in size and is apparently present only intermittently. Plants and animals inhabiting the seep could experience toxic exposures. However, because the stock tank and seep can attract wildlife, exposure by organisms seeking drinking water to elevated levels of constituents in these surface waters is enhanced.

7.3.1.3 Ecological endpoints

Reference toxicity values for many of the pollutants were obtained from EPA documents on sludge toxicity and from EPA Water Quality Criteria. Little toxicity information is available for sagebrush ecosystem wildlife and plants. Surrogate values for comparison were obtained therefore, from available studies. These studies often involved laboratory species. Extrapolation from these studies to Silver Mountain Mine site is a source of uncertainty as discussed in the Toxicity Assessment.

7.3.2 Exposure pathways

Chapter 5 reviews in detail the fate and transport of contaminants.

Present condition with heap cover intact.

Apart from the mine drainage to the stock tank, transport pathways appear to be minimal. The air pathway is negligible due to size of the heap particles and presence of the cover. Surface water pathway is minimal due to the cover which prevents leaching and runoff from the heap (but not the dump or nearby soils) and due to the lack of rainfall which minimizes the presence of surface waters throughout most of the year. Groundwater transport is apparently low. As long as the cover is in place and prevents further leaching, no further export of metals from the heap is expected, although the dump, nearby soils, and the mine drainage will continue to be sources to ground water and intermittent surface waters.

The liner probably affords little protection against leachate seepage. The leachate would collect in the impoundment pond to evaporate or leak to the ground.

The air pathway is likely to be insignificant. Modeling analysis of the existing leach heap (Appendix M), indicates that there will be no emissions of particulates from the pile under existing conditions. Analysis in Chapter 6.3.5.1.2 indicates that only low concentrations (0.031 to 0.53 $\mu\text{g}/\text{m}^3$) of hydrogen cyanide will volatilize to the atmosphere, using worst-case assumptions.

With the cover intact, exposure to the heap, particularly the south toe, will probably not occur. However, the dump has high arsenic toxicity (as does the nearby soil) to vegetation and ruminants. In addition, phytotoxicity is possible from selenium concentrations. Toxicity due to manganese concentrations is also possible, and rodents and birds could be exposed to toxic selenium concentrations. Zinc is also likely to be toxic (but probably not lethal) to birds.

Future condition when cover fails

When the cover no longer prevents exposure to the heap, then the heap, and its very toxic south toe will become available to use by plants and wildlife. As discussed in section 7.3.6, below, the likelihood of such use is limited for the heap soils in their present condition (steep pile of large-sized rocks).

7.3.3 Exposure point concentrations

Concentrations of pollutants were measured in soil, surface water, and ground water samples. These were compared directly, where possible, with reference values.

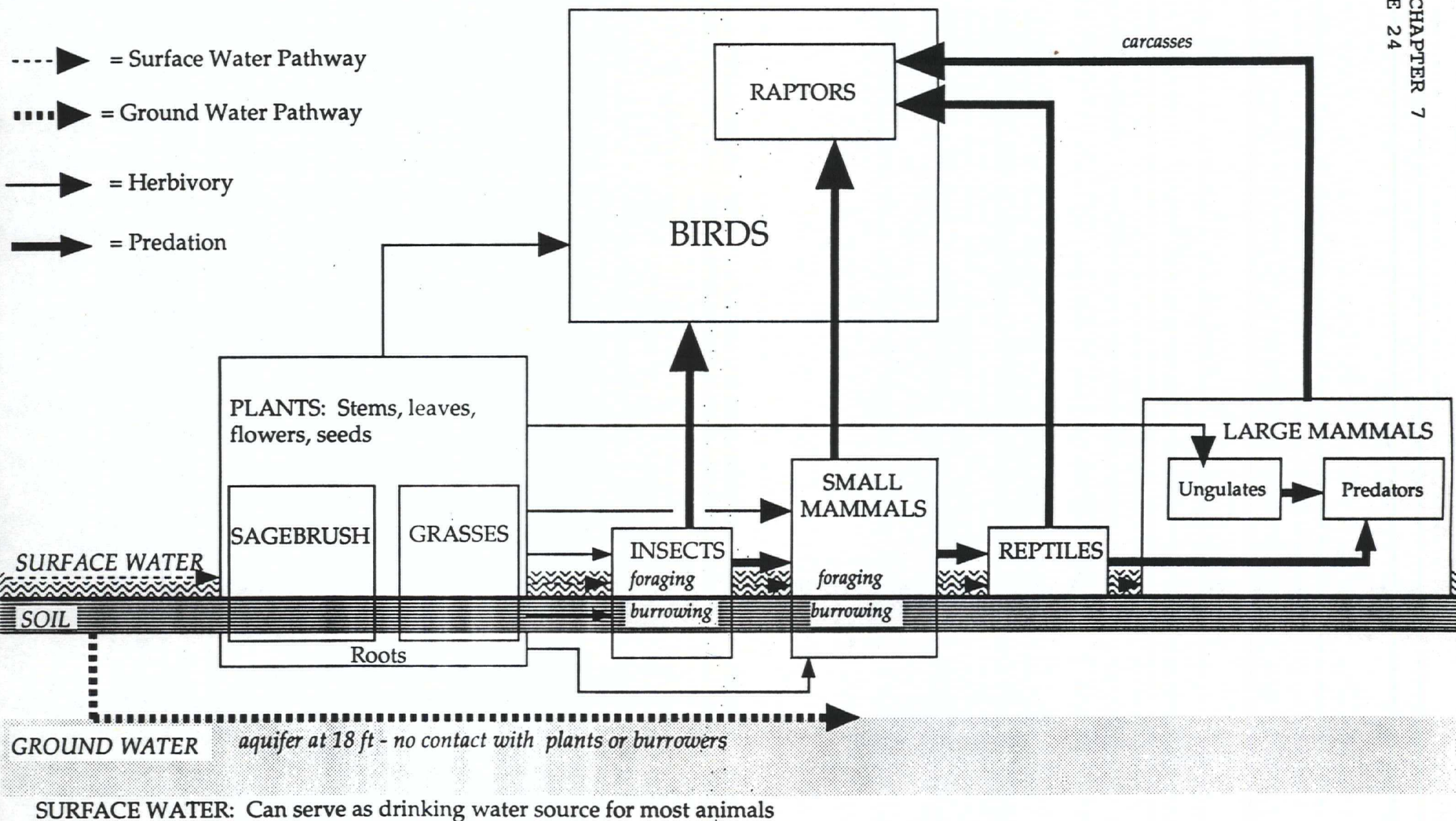
For ground water samples exhibiting toxicity, a dilution of 800 was applied (see Chapter 5.2.2) to estimate the exposure point concentrations where the ground water is most likely to enter surface waters.

7.3.4 Site model

The site model, depicting sources, types of pollutants, affected media, routes of exposure, and endpoints, has been presented previously (Figure 6.1). Figure 7.1 shows more detail of the interaction of the plant and animal components within the

SILVER MOUNTAIN MINE CONCEPTUAL MODEL

FOOD WEB CONNECTIONS



sagebrush ecosystem, based on the description presented earlier in section 7.3.1.1.

7.3.5 Exposure summary

Soil. Only a limited possibility exists for the pollutants in soils to affect wildlife. Because of the lack of transport and subsequent exposure, the population most exposed at present is the burrowing fauna utilizing the heap, the nearby soils and the dump. This is probably an extremely limited fauna since the heap and dump particles are not the equivalent of surface soils and both piles are relatively small in size. With the cover on the heap, accumulation of soil and subsequent plant growth on the heap is unlikely, so herbivores are probably not at risk from the toxic concentrations in the heap soils. It is uncertain whether plants, insects, or rodents can successfully utilize the dump. These organisms form the food base for herbivores and predators, so their absence at the dump will minimize possible toxic effects on ruminants and raptors. If the heap (or dump) were flattened out and subject to soil formation processes, the likelihood of the soils exhibiting toxicity would increase greatly.

Water. Exposure of wildlife to ground water will occur when the ground water reaches surface waters. At this point the groundwater will be diluted sufficiently to reduce toxicity below reference levels. Existing surface waters act as attractants to wildlife. In particular, water that is drained from the mine to the cattle trough enhances exposure.

7.3.6 Uncertainty

Quantitative estimates of exposure are lacking. Anecdotal information suggests that any surface waters will be utilized by on-site fauna. Densities of organisms on site are unknown, but several species have been observed to utilize the site, and regional experts have provided a list that covers at least the bird species of concern in the area (Table 7.1).

7.4 RISK CHARACTERIZATION

Given the existing condition of the heap, there is essentially no threat to surrounding wildlife from the heap because (1) the existing cover will result in very little if any leachate formation, and (2) whatever leachate forms will enter the ground and would not be available as a surface drinking water source. There is some possibility of ponding occurring over an

unknown, but likely short, period of time following a heavy rainfall.

Wildlife have unrestricted access to the dump and nearby soils. Those organisms at risk from soils would be the ones actually inhabiting the heap or dump. This would be an extremely limited population of burrowers perhaps including some of the insects and small mammals identified previously. When the cover on the heap fails, this same suite of species will be exposed to the heap. Plants are unlikely to be exposed to heap or dump soils.

The greatest existing risk to wildlife and plants appears to be from the arsenic concentrations in the nearby soils. These soils are likely to be utilized by sagebrush biota, although the area involved is small. The nearby soils are contaminated with levels of arsenic toxic to vegetation and ruminants, levels of manganese possibly toxic to rabbits, levels of selenium that may be toxic to rodents and birds, and levels of zinc that are likely to be toxic to birds feeding on soil biota. In the future, once the heap cover deteriorates, there may be some acute toxicity at times from temporary ponding of leachate. Soils from the heap and dump may exert their potential toxicity if they erode, spread out, leach, etc.

There is no current risk to wildlife or plants from ground water, and no future risk is anticipated. Surface waters, however, attract wildlife, enhancing exposure to toxic levels of pollutants within those waters. The mine drainage to the trough will probably continue to be a source of elevated arsenic concentrations. To a lesser extent, the seep area may continue to be a source of elevated aluminum, copper, and lead.

7.5 SUMMARY

Research has demonstrated that wildlife in semi-arid areas are attracted to surface waters. Any continued leachate that collects could again cause local acute effects on wildlife.

Because the pad liner has deteriorated, collection of leachate now or in the future is unlikely. It is assumed that the leachate will enter the ground water without ponding (except perhaps during heavy storms) and will therefore not be a source of acute toxicity.

The cover on the heap will prevent leachate from forming and restrict access to the heap by large organisms. The cover will also prevent dispersal of any soils from the heap. Because the

heap is not the equivalent of soil (due to large particle sizes and likely absence of a normal sagebrush soil biota), the density and types of organisms that will utilize the heap is likely to be minimal. Once the cover deteriorates, leachate may be able to form during wet periods. At that time, vegetation, wildlife, and other biota will be potentially exposed to toxic concentrations of metals in the heap. Soils from the heap and dump are most likely to exert their potential toxicity if they erode, spread out, leach, or otherwise made more available to on-site biota.

Air transport of particulates from the leach heap is negligible. Ground water is not toxic to plants or aquatic biota at present. Surface water transport is absent for most of the year. Transport to these nearby sensitive communities is negligible. The nearby soils are contaminated with arsenic, manganese, selenium, and zinc, that can affect vegetation, ruminants, rabbits, rodents, and birds.

REFERENCES

- Carson, R.G. and Peek, J.M. 1987, Mule deer habitat selection patterns in northcentral Washington: J. Wildl. Manage., v. 51, no. 1, p. 46-51.
- Clark, J.M. 1988, Ecological risk assessment at Crab Orchard National Refuge: A Superfund Site: US Environmental Protection Agency, Region V.
- Denton, D.K., and Benham, J.R. 1989a, Report of findings and remedial recommendations on the Silver Mountain Mine and mill tailings, Okanogan County, Washington: US Bureau of Mines, Western Field Operations Center, Spokane, WA.
- Denton, D.K., and Benham, J.R. 1989b, Report of findings and remedial recommendations on the Silver Mountain Mine and mill tailings, Okanogan County, Washington. U.S. Bureau of Mines, Western Field Operations Center, Spokane, WA. June 27, 1989 version.
- Eberhardt, L.E., Hanson, E.E., and Cadwell, L.L. 1984, Movement and activity patterns of mule deer in the sagebrush-steppe region. J. Mamm., v. 65, no. 3, p. 404-409.
- Eisler, R. 1988, Arsenic hazards to fish, wildlife, and invertebrates: A synoptic review: U.S. Fish Wildl. Serv. Biol. Rep. v. 85, no. 1.12, 92 pp.
- Kapustka, L.A. 1989, Letter of June 30, 1989, to EPA Region 10. Analysis of Aerial Photographic Analysis of the Silver Mountain Mine Area, Okanogan County, Washington.
- King, J.B. 1989 Letter, containing species list, from Jerry King, State of Washington, Department of Wildlife, Region 2, Ephrata, WA, to P.B. Duncan, USEPA, Region 10, Seattle, WA, 10/27/89.
- Knight, R.L. and Smith, D.G. 1982, Summer raptor populations of a Washington coulee: Northwest Science v. 56, no. 4, p. 303-309.
- McCorquodale, S.M. 1987, Fall-winter habitat use by elk in the shrub-steppe of Washington: Northwest Science v. 61, no. 3, p. 171-173.

-----, Raedeke, K.J. and Taber, R.D. 1986, Elk habitat use patterns in the shrub-steppe of Washington: J. Wildl. Manage. v. 50, no. 4, p. 664-669.

National Academy of Sciences 1977a, Arsenic: National Research Council. Committee on medical and biologic effects of environmental pollutants: Washington, D.C. 332 pp.

----- 1977b, Copper: National Research Council. Committee on medical and biologic effects of environmental pollutants: Washington, D.C. 115 pp.

----- 1973, Manganese: National Research Council. Committee on medical and biologic effects of environmental pollutants: Washington, D.C. 332 pp.

Rickard, W.H. and Garland, T.R. 1983, Trace element content of leaves of desert shrubs in south-central Washington: Northwest Science v. 57, no. 1, p. 57-61.

-----, Rogers, L.E. 1983, Industrial land-use and the conservation of native biota in the shrub-steppe region of western North America: Environmental Conservation v. 10, no. 3, p. 205-211.

-----, Van Scoyoc, S. 1984, Trace element uptake by sagebrush leaves in Washington: Environmental and Experimental Botany v. 24, no. 1, p. 101-104.

Schwartz, S. 1983, Nature in the Northwest: Prentice-Hall, Inc., Englewood Cliffs, N.J. 256 pp.

Sheldon, J.K. and Rogers, L.E. 1978, Grasshopper food habits within a shrub-steppe community: Oecologia v. 32, p. 85-92.

Svedburg 1989, Personal communication (07/18/89) with Svedburg of the Department of Wildlife concerning species of concern in the area.

US Bureau of Mines 1988, U.S. Dept of Interior, Bureau of Mines Letter from William B. Schmidt to Keith Rose, EPA Region 10, July 28, 1988.

US Environmental Protection Agency 1985a, Environmental profiles and hazard indices for constituents of municipal sludge: Office of Water Regulation and Standards.

Arsenic	EPA 0215
Beryllium	EPA 0220

Cadmium	EPA 0222
Chromium	EPA 0228
Cobalt	EPA 02--
Copper	EPA 0230
Iron	EPA 0239
Lead	EPA 0260
Mercury	EPA 0242
Molybdenum	EPA 0246
Nickel	EPA 0247
Selenium	EPA 0252
Zinc	EPA 0259

----- 1985b, National Primary Drinking Water Regulations:
Synthetic organic chemicals and microorganisms; Proposed
rule: Federal Register, Vol. 50, No. 219, November 13,
1985.

----- 1985-1987, Ambient water quality criteria: Office of
Water Regulation and Standards:

Ammonia	EPA 440/5-85-001	1985.
Arsenic	EPA 440/5-84-033	1985.
Cadmium	EPA 440/5-84-032	1985.
Chromium	EPA 440/5-84-029	1985.
Copper	EPA 440/5-84-031	1985.
Lead	EPA 440/5-84-027	1985.
Nickel	EPA 440/5-86-004	1986.
Zinc	EPA 440/5-87-003	1987.

----- 1989, Aerial Photographic Analysis of the Silver Mountain
Mine Area, Okanogan County, Washington: Environmental
Monitoring Systems Laboratory, Las Vegas. TS-PIC-89726.
April 1989. 17 p.